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STREET LANDFILL SITE STUDY,
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HYDROGEOLOGICAL INVESTIGATIONS OF
THE UPPER OTTAWA STREET LANDFILL SITE

A Progress Report For the Period:

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PREFACE

This report consists of two parts. Part 1 describes the design, development and applications of a device for monitoring water levels and water quality at numerous depths in a single borehole in fractured rock. This device was developed by the University of Waterloo to provide an efficient method for monitoring groundwater conditions at the Upper Ottawa Street landfill site. Part 1 was prepared as a paper that has recently been submitted (without Part 2) to the Canadian Geotechnical Journal for peer review and possible publication in late 1983.

Part 2 describes the hydrogeological and hydrochemical investigations that have been conducted by the University of Waterloo at the Upper Ottawa Street landfill site in 1982. It includes data obtained in 1982 and in early 1983. Much of the chemical data presented in Part 2 were produced by various analytical laboratories in late 1982 and early 1983. Some of the groundwater samples collected in 1982 are still in the process of being analysed. The data tabulations and interpretations in Part 2 are therefore preliminary. This investigation represents the first attempt in Canada to delineate in detail a major zone of contamination at a large waste disposal site on permeable fractured rock in a complex topographic setting. Concurrently with the site investigation, it has been necessary to develop and assess the new method for multilevel groundwater monitoring. Part 1 of this report

describes the multilevel monitoring system in the context of its usefulness for monitoring specifically at the Upper Ottawa Street landfill site and secondarily at sites on other types of fractured rock in Ontario where the device has been applied on an experimental basis. These applications are part of other hydrogeological research projects being conducted by the University of Waterloo with funding provided from sources other than the Upper Ottawa Street Landfill Site Study.

Investigations by the University of Waterloo of the hydrogeological conditions at the Upper Ottawa Street landfill site began in the fall of 1981. Initially, the existing hydrogeological information on the site was reviewed to assess the potential for a detailed site study using conventional methods of groundwater monitoring. It was concluded that because of the varied topography and because of the expected occurrence of a complex network of groundwater flow paths through fractures in the dolomite and shale at the site, detailed monitoring of the groundwater system would require monitoring at many depth levels and at many locations. It was recognized that if conventional standpipe piezometers or observation wells were to be used to create such a monitoring network, the costs for drilling the large number of holes necessary would be prohibitive even in a project with an exceptionally high level of funding.

A review of the literature pertaining to alternative methods of groundwater monitoring in fractured rock was undertaken. By combining and modifying monitoring concepts described in the literature and with the aid of an unusual chemical sealant material that, for many years, has been used for sealing deep boreholes at nuclear test sites in the

The first section of the document discusses the importance of maintaining accurate records of all transactions. It emphasizes that proper record-keeping is essential for the integrity of the financial system and for the ability to detect and prevent fraud. The text also mentions the need for regular audits and the importance of having a clear chain of custody for all documents.

The second section describes the various methods used to collect and analyze data. It includes a detailed explanation of the sampling process and the use of statistical techniques to interpret the results. The text also discusses the importance of ensuring that the data is representative of the population being studied and that the analysis is unbiased.

The third section outlines the procedures for conducting field research. It provides a step-by-step guide to the process, from the selection of the study area to the collection of data. The text also discusses the importance of obtaining the necessary permits and the need to follow ethical guidelines when working with human subjects.

The fourth section discusses the results of the study and the conclusions that can be drawn from the data. It includes a detailed analysis of the findings and a discussion of the implications for the field. The text also mentions the need for further research and the importance of sharing the results with the scientific community.

The final section of the document provides a summary of the key points and a list of references. It also includes a brief discussion of the limitations of the study and the need for further research.

United States, a preliminary design for the multilevel monitoring device described in Part 1 of this report was conceived. In November, 1981, a prototype of the device was installed in an 18 m deep borehole at the landfill site. In December, 1981, we outlined a proposed hydrogeological investigation of the Upper Ottawa Street landfill site for the 1982-1984 period. This investigation began in a preliminary form in April. The intensive stage of the investigation was initiated in July when funding was approved by the Province of Ontario.

HYDROGEOLOGICAL INVESTIGATIONS OF
THE UPPER OTTAWA STREET LANDFILL SITE

PART 1

DEVELOPMENT AND APPLICATIONS OF
A MULTILEVEL DEVICE FOR GROUNDWATER MONITORING
IN FRACTURED ROCK

ABSTRACT

A device has been developed for monitoring hydraulic head and for groundwater sampling at many depths in a single borehole in fractured rock. The device is constructed from readily-available materials and it can be installed manually in open boreholes to depths of 100 m or more. Relative to the total cost of drilling for conventional piezometer nests that require several boreholes to different depths at each monitoring site, the costs of the device and the time required for installation after the hole is drilled are low. The device requires only simple inexpensive equipment for water-level monitoring and for water sampling. The device is a permanent installation in the borehole in the same manner as a conventional well or piezometer.

The multilevel device consists of a number of piezometer tubes, which are normally 9 mm or 12 mm I.D. polypropylene, polyethylene or Teflon tubes, contained within a PVC casing. To form a piezometer port for access to the groundwater domain outside the PVC casing, each tube is connected to a 90⁰ plastic or nylon elbow that is sealed in a hole in the casing. Each piezometer tube extends to an elbow at a different depth in the casing. Above and below each piezometer port the borehole is sealed by packers consisting of a cylindrical sealant sleeve with a rubber cover that is secured to the casing at the top and bottom of the packer. The sealant sleeve is composed of a chemical compound (Dowell sealant) that expands in the presence of water that flows through holes

in the casing segment covered by the packer. The rubber sleeve prevents chemical reactions between the sealant and the formation water outside the casing. As the assembled multilevel device is lowered into the borehole, water is poured into the casing to overcome buoyancy and to inflate the packers. Inflation occurs slowly over a period of several hours. After inflation, the water in the casing can be replaced by a slurry of bentonite. This prevents long-term dissolution of the sealant and ensures that there is no leakage at couplings or at the elbow joints.

Several prototypes of the multilevel device have been tested in the laboratory. Devices with various modifications have been installed in twenty 7.5 cm diameter vertical boreholes at a hazardous waste disposal site on fractured dolostone and shale, in six 11 cm diameter vertical boreholes at a municipal landfill on fractured shale and in one vertical hole and three angle holes (7.5 cm diameter) in fractured metasedimentary Precambrian rock at a uranium tailings impoundment. In the 11 cm diameter holes the device has a maximum of 12 piezometers and in the 7.5 cm diameter holes it has a maximum of 6 piezometers. As these field investigations proceeded, deficiencies in the design, construction and installation procedures were recognized and modifications were made and then evaluated in later installations. The most recent version of the device provides convenience of assembly and installation in the field and preliminary field data indicate that the monitoring performance is good. It is expected, however, that as more experience with the device is acquired in different field situations, additional modifications will be made to provide greater ease of construction and reliability.

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INTRODUCTION

This paper describes recent modifications and applications of a multilevel device for groundwater monitoring in fractured rock. An early version of the device was described by Cherry and Johnson (1982). The device, which consists of many narrow diameter piezometers in a single casing, is a simple and inexpensive means for multilevel monitoring in open boreholes. In this context, an open borehole is a hole that, after the drill rods and bit are removed, remains open for at least an hour or two so that the device can be installed. The device is a permanent installation in the hole rather than one that is lowered into the hole for a series of tests and then removed. Although many possibilities exist, the devices have been produced in sizes suitable for only three sizes of boreholes. An 11 cm hole takes a device that has a maximum of about 12 piezometers, a 3.5 cm hole takes a device that has a maximum of 9 piezometers, and a 7.5 cm hole takes a device that has a maximum of about 6 piezometers.

The multilevel device was developed in response to a need to determine zones of contaminant migration at waste disposal sites on fractured rock in which the distribution of fractures is generally complex. In these types of environments, identification of fracture zones where contaminated groundwater occurs often necessitates the acquisition of data from many depth intervals at each monitoring site. The multilevel device appears to be an alternative to the conventional

well nest and its relatively high cost of drilling. Within a single borehole the device provides many separate monitoring intervals for which individual hydraulic-head measurements and water samples can be obtained.

The monitoring device has the following capabilities. The intervals for monitoring in the borehole can be distributed by the choice of packer spacing so that nearly the entire length of the borehole is monitored in discrete segments, or so that smaller segments comprising only a portion of the total borehole can be monitored. Because the piezometer tubes are narrow and because the borehole annulus around the piezometer tip has a small volume and no sand pack, water samples can be drawn from the formation without having to remove large volumes of water to entirely flush out the piezometers. Even in low-permeability deposits the response times for the piezometers are not long because the narrow diameter of the piezometer tubes provides small storage volumes.

In conventional piezometers or observation wells, the water in the sand pack around the screen is in contact with the bentonite or grout seal above the pack. Bentonite and grout, which are relatively reactive, may influence the chemical composition of the water drawn through the sand pack into the piezometer, particularly in low permeability materials where the flow through the sand pack is slow. The seals used in the multilevel device do not present this problem because they are isolated by a rubber membrane from the water in the sample zones.

The installation of the device in the borehole can be accomplished manually by one or two persons or with the aid of the winch and mast on the drill rig. Once the device is in place, conventional narrow-diameter electric probes are used to monitor water levels in the piezometers and water samples may be collected using simple and inexpensive methods.

The initial version of the monitoring device was developed in 1981 and two prototypes were constructed and subjected to preliminary tests in the laboratory. A full-scale assembly was then installed in a vertical borehole in fractured Paleozoic dolomite and shale at a hazardous waste disposal site in the City of Hamilton, Ontario. This demonstrated rapid assembly and easy installation in the field. In 1982, nineteen more of these devices were installed in vertical holes at the site in Hamilton, six were installed in vertical holes in fractured Paleozoic shale at a landfill in the City of Burlington, Ontario, and three were installed in angle holes in fractured metasedimentary Precambrian rock at a tailings impoundment in the Elliot Lake uranium district of Ontario. As the field investigations continued, the construction and the installation and the overall performance of the device were assessed. This assessment, in addition to the results of laboratory studies on construction materials and methods of assembly, identified various deficiencies in the device. Design modifications and the rationale for them are presented in this paper.

The descriptions of materials and construction details provided in this paper represent a progress report. Although some of the details will continue to evolve as more field and laboratory experience is

acquired, it is unlikely that the basic components of the design will undergo much change in the near future.

CONCEPTUAL DESIGN

The device, which is displayed schematically in Figures 1 and 2, consists of a bundle of piezometer tubes contained within a casing (i.e. pipe) that is capped at the bottom. Each of the piezometer tubes is connected to an intake port that consists of a 90⁰ elbow that protrudes through the casing. The elbow is sealed in a hole in the casing so that there is no flow of water between the outside and the inside of the casing. If sediment from the formation is expected to be a problem, the intake port (i.e. the open end of the elbow) can be covered with a screen or permeable fabric.

For samples from each piezometer to represent the conditions that exist in the formation, it is essential that the borehole intervals above and below each intake port be sealed. This sealing of the borehole above and below each port is accomplished by means of self-inflating cylindrical packers that consist of a soft rubber-like sealant (Dowell sealant) composed primarily of organic compounds. The sealant is a patented material manufactured by the Dow Chemical Company.

When the sealant is contacted by water, it expands slowly to fill the borehole annulus. Because the sealant has appreciable solubility of organic compounds, it is desirable for groundwater quality monitoring that it not be in contact with water in the borehole annulus. To maintain the chemical integrity of the groundwater samples, the sealant sleeve is isolated from the groundwater by a rubber sleeve that is secured tightly to the casing above and below the Dowell sealant sleeve.

In order to allow water to contact the sealant, holes are drilled in the segments of casing on which the Dowell sealant sleeves are fitted. After the device is placed in the borehole, water poured into the casing passes through these holes to contact the sealant. This added water also serves to overcome the buoyancy effect that occurs when a capped, water-tight casing is lowered down a water-filled borehole. The Dowell sealant sleeve absorbs water and, during a period of several hours, it swells until it presses against the borehole wall. It continues to swell for many days until a high swelling pressure is achieved.

It is desirable that one of the tubes within the casing be open-ended, rather than be attached to an elbow to form a piezometer port. This tube should extend nearly to the bottom of the casing. It has three purposes. It provides (i) an access route for measuring the water level in the casing, (ii) a means for injecting dye or bentonite slurry into the casing and (iii) a means for pumping water out of the casing. Pumping from the casing with subsequent water-level monitoring in the piezometers and dye injections are done in order to test the casing for leaks. The injection of bentonite slurry into the casing is a means of sealing the casing in the event that leakage is detected in these tests. It also serves to prevent long-term dissolution of the Dowell sealant.

The multilevel device can be used to monitor short discrete zones or long borehole segments (Figure 1). For discrete-interval monitoring, the packers above and below each piezometer port are close together. Discrete-level monitoring can provide hydraulic-head values and water samples representative of specific fractures or fracture zones. When the monitoring intervals are short, and when only a small length of

borehole is occupied by the packers above and below each piezometer tip, large intervals of the borehole annulus between monitoring zones remain open.

In situations where the rock has a large number of fractures, segments of open hole without packers are undesirable because they may allow vertical flow (i.e. short circuiting) along the borehole. This flow may be caused by vertical hydraulic-head differentials between one fracture and another or between fracture zones. This flow disturbance between the monitoring points may influence the monitoring results. In some cases it is possible that when a piezometer is pumped, water in the borehole annulus above or below the packers will be drawn around the packers via fractures and into the piezometer. The possibilities for such influence also introduces uncertainty with respect to the integrity of the monitoring data.

This uncertainty can be avoided if the intervals between monitoring zones are fully sealed by use of additional packers. If many packers are required to fill these intervals, the cost of the extra packers can greatly increase the total cost of the monitoring device.

An alternative approach for sealing the intervals between the monitoring zones is to use some of the piezometer tubes for injection of cement grout or bentonite slurry into the intervals. These tubes would then not serve as piezometer tubes. The maximum number of piezometers in the multilevel device would therefore be about half of what would otherwise be the case.

Hand-operated vacuum pumps or peristaltic pumps are a convenient means of obtaining samples from the piezometers when the water levels

are less than about 8 m below ground surface. If the water level is deeper, water samples can be obtained by gas-lift pumping using an air compressor or nitrogen tank, by means of a bailer consisting of a narrow tube with a check valve on the bottom or by means of a triple-tube gas-drive sampler of the type described by Robin et al. (1982). Although the piezometer tubes are narrow, it is normally possible to obtain adequate sample volumes for analysis of nearly all constituents of interest in water-quality investigations.

PERFORMANCE OBJECTIVES AND ASSESSMENT

Each of the components of the device must meet one or more performance objectives. The monitoring tubes must have a sufficiently large diameter for water levels to be conveniently measured using an electric water-level probe and be large enough to prevent measurement errors due to capillary rise along the tubes. In areas where water levels are deep, the tubes should also be large enough for use of narrow-diameter pumps such as those described by Robin et al. (1982). The diameter of the tubes should be sufficiently small so that many tubes can be fitted inside the casing. These objectives are generally achieved when tubes having inside diameters of 9 mm (3/8 inches) or 12.5 mm (1/2 inches) are used.

The size of borehole best suited for a particular monitoring task depends on factors such as drilling costs, the number of monitoring points desired, the standard sizes of pipe and couplings available for construction of the casing and the relative ease with which the assembly can be handled during construction and installation. The packers are pre-assembled and are brought to the field as casing segments that are joined to blank lengths of casing pipe or lengths of casing pipe into which the piezometer ports are inserted. In order to minimize drill rig costs, it is desirable that the monitoring device consist of various components that can be easily assembled on site and installed without the use of a drill rig.

If no leakage occurs after the multilevel device is installed in the borehole and after the packers are inflated, the water level of the standing water column in the casing will not vary. There are three potential sites where leakage between the outside and the inside of the casing may occur; at improperly joined couplings, at improperly sealed piezometer elbows, or at the clamps on the rubber sleeves on the packers. When fluctuation in the level of the standing water column in the casing indicates that leakage occurs, it may be necessary to determine which sampling point or points are influenced by the leakage. This can usually be accomplished by pumping a dye tracer down through an open tube that extends close to the bottom of the casing.

Another means of determining the sampling zone or zones that are influenced by leakage is to raise or lower the water level in the casing and monitor the water levels in each of the piezometer tubes. It is conceivable, however, that a borehole zone that has a very high hydraulic conductivity would not show a detectable water level change if the leakage rate is small relative to the rate at which water flows into or out of the fractured rock. In this situation even dyes or other chemical tracers may not provide evidence of leakage when the leakage water is strongly diluted by the formation water in the sampling zone. The uncertainties associated with leakage can generally be avoided if the water column in the casing is replaced with a bentonite slurry after the packers are inflated. This is accomplished by mixing powdered expandable bentonite with water to a consistency that can be pumped down a narrow (10 mm) open tube. The input of the slurry at the bottom of the casing causes displacement of the water column. The slurry

gradually forms a nearly solid mass within the casing as the bentonite expands. In addition to sealing leakage points, the bentonite slurry essentially eliminates the possibility for appreciable quantities of sealant-derived chemicals to enter any of the sampling zones. It prevents hydraulic communication within the casing between one level and another. All of the piezometer ports that have good elbow joints should therefore yield water-level data that are uninfluenced by any previously existing imperfect seals at couplings or at packer clamps.

Except in rare cases where there is a major break in the casing or where a piezometer tube is detached from its elbow, the placement of the bentonite slurry in the casing is a means of overcoming the uncertainties that arise when water-level tests in the casing indicate leakage. It could be argued therefore that, rather than filling the casing with water when it is installed, the casing should be filled initially with a bentonite slurry. This would inflate the packers and it would prevent outflow of sealant-contaminated water into the formation if leakage from the casing occurs. The advantage of having a water-filled casing for the initial period is that water-level tests can be easily used to determine whether or not leakage occurs. If it is shown that the casing is watertight, which should be the case with few exceptions, there is no uncertainty with regard to hydraulic or contaminant communication between the interior of the casing and the borehole. Leakage occurred in most of the multilevel devices installed at the Hamilton and Burlington field sites. No leakage occurred in any of the devices installed at the Elliot Lake field site. The devices used at this site contain several design improvements made as a result of the experience gained at the other two sites.

MATERIALS, CONSTRUCTION AND INSTALLATION

All of the multilevel devices that have been assembled, both as prototypes and for field installations, have had casings constructed of PVC pipe, threadless PVC couplings, and piezometer tubes made of polyethylene or polypropylene tubing connected to polyethylene or nylon elbows. The coupling and pipe sizes are selected so that the assembly prior to packer inflation will fit loosely in the borehole. In boreholes with a diameter of 7.6 cm (3 inches) standard 5.1 cm inside diameter (nominal 2 inch ID) Schedule 40 PVC pipe is used, in 9.5 cm boreholes standard 6.1 cm inside diameter (Nominal 2.5 inch ID) Schedule 40 PVC pipe is used, and in 11.4 cm (4.5 inch) boreholes standard 7.7 cm (nominal 3 inch ID) Schedule 40 PVC pipe is used. The outside diameter of the couplings determines the maximum outside diameter of the sealant sleeves with their rubber covers. The diameter of the couplings should actually be slightly larger than the outside diameter of the rubber-sleeve on the packer. This prevents the packers from being abraded or punctured by contact with the borehole wall during installation. Standard Schedule 40 PVC couplings for 5 cm, 6.1 cm and 7.6 cm ID PVC pipe have outside diameters of 7.2 cm, 8.6 cm, and 10.3 cm, respectively. The clearances in 7.6 cm, 9.5 cm, and 11.4 cm diameter boreholes are therefore about 2, 5 and 5 mm respectively.

A detailed representation of a packer is shown in Figure 2. The Dowell sealant sleeve is placed over a layer of cloth (210 mesh

monofilament nylon screen) that is attached by tape to the PVC pipe. This single layer of cloth facilitates wetting of the Dowell sealant.

The rubber sleeve that covers the Dowell sealant sleeve should be sufficiently strong to provide good durability and sufficiently dense and thick to minimize the passage by diffusion of chemical constituents through the sleeve. The rubber sleeve must also be sufficiently expandable so that the swelling pressure exerted by the Dowell sealant will press the sleeve strongly against the borehole wall. In the early version of the device described by Cherry and Johnson (1982), the sleeve consisted of a very thin rubber membrane of the type normally used in triaxial cell tests in soil engineering laboratories. In the recent versions of the device, a much thicker and much less expensive rubber is being used. The rubber sleeve is the same material that forms conventional rubber bands used in offices or schools. The thickness of the rubber sleeve used on the packers in the 11 cm boreholes is 2.0 mm and on the packers in the 7.5 and 9.5 cm boreholes the rubber is 1.0 mm thick.

As the multilevel device is placed in the borehole, water is added to the casing to maintain a slight positive head differential between the inside of the casing and the water level in the borehole. The rubber sleeve should be tightly clamped to the casing so that water that has been in contact with the Dowell sealant does not flow along the outside of the casing into the borehole. The head differential should not be so large as to cause hydraulic rupture of the rubber sleeves. Once the Dowell sealant expands, the tightness of the clamp on the rubber sleeve is of lesser importance.

In the device described by Cherry and Johnson, (1982) the rubber sleeve was attached to the casing by an O-ring set in a narrow groove. This method of attachment was found to be insufficiently tight to prevent leakage of water across the groove. A more reliable method for securing the rubber sleeve is provided by plastic or metal strap fasteners tightened in grooves in the PVC pipe. Double fasteners in double grooves at each end of the packer with silicone sealant in between the double grooves provide a reliable seal. An alternative method for securing the ends of the rubber sleeve is to bevel and groove the ends of the packer pipe, extend the rubber sleeve to the end of the pipe and force the coupling tightly over the end of the rubber sleeve. In this manner the rubber sleeve is secured simply by the coupling without the use of fasteners or clamps.

The Dowell sealant sleeve is produced by pouring a liquid mixture of the chemical components that form the sealant into a mold and allowing the mixture to solidify overnight. Two types of molds are being used. The first type is simply an outer aluminum tube and a piece of the PVC casing used as the inner cylinder. The two tubes are set in a grooved base to maintain proper positioning. After the sealant sleeve has solidified, it is extruded from the mold by pressure exerted by a jack. To facilitate the removal process, the walls of the mold are initially sprayed with silicone lubricant. By this method it is possible to make sealant sleeves that have a maximum length of approximately 30 cm. Beyond this length the extrusion process commonly damages the sleeve.

The second type of mold is one in which the outer tube consists of two half-cylinders of aluminum pipe joined along one edge by hinges and by clamps along the other edge. The sleeve is easily removed when the clamps are detached. The sealant sleeve can then be peeled off the inner PVC tube. In this manner sleeves that are a metre or more in length can be made.

In the early version of the multilevel device described by Cherry and Johnson (1982), the elbow was attached to the PVC casing without the use of solvent joiner. An O-ring was inserted over the threaded end of the elbow where it extends through to the outside of the pipe. A threaded nylon nut was tightened on the O-ring to compress the elbow to create a water-tight seal. This method of securing the elbow to the casing was discontinued because it provides uncertainty with regard to the tightness of the join. In the method used at present the elbow is threaded with Teflon tape into a short length of PVC pipe and the pipe is solvent jointed into the casing. This prevents leakage, but it introduces solvents into the system, which can be undesirable when monitoring some types of trace organic compounds in groundwater.

In situations where monitoring for trace levels of organic compounds in groundwater is required, it may be appropriate to construct the multilevel device of materials that will not give off or adsorb organic compounds. This results in a large increase in cost of the multilevel device. In place of polyethylene or polypropylene tubing, Teflon tubing or Teflon-lined polyethylene tubing can be used for the piezometers. Stainless steel pipe can be used for the casing segments to which the piezometer ports are attached. Rather than use stainless

steel for the entire casing, the steel segments that contain piezometer tips can be attached by threaded couplings with Teflon tape seals to the packers above and below the monitoring interval. The avoidance of long lengths of steel casing facilitates handling in the field and it reduces the cost of the assembly.

The multilevel device can be assembled quickly at the borehole by a two or three-person crew. The spacing of packers, blank sections and monitoring intervals are specified after the hydrogeologic conditions of the borehole are assessed by means of core inspection and/or borehole tests. The desired spacing of packers and sampling ports is achieved by cutting appropriate lengths of PVC pipe in the field or by use of various pre-cut lengths of pipe complete with sampling port holes.

The multilevel device is assembled in sections laid out on the ground near the borehole. Lengths of tubing for the piezometers are uncoiled and cut off to the desired piezometer lengths. The 90° elbows are joined to the piezometer tubes by insertion into the ends of the tubes. The integrity of these joins can be increased by use of steel O-clamps. A short length of PVC pipe is threaded onto the other end of the elbow so that the elbow can be glued into the sampling port hole in the PVC casing.

The port elbow for the deepest piezometer is then inserted into the hole in the bottom casing segment. This casing segment is then lowered into the upper part of the borehole. The top of this segment is held just above ground level while the next section of casing is raised and coupled to it. After the solvent joiner has been allowed to set for several minutes, the casing segments are lowered farther into the hole.

This procedure is repeated until the entire assembly is in the borehole. When each length of casing is inserted into the borehole, the piezometer tubes are bound together with fiberglass tape. This procedure prevents twisting of the piezometer tubes within the casing thereby facilitating water-level measurement and water sampling.

FIELD APPLICATIONS

A Closed Landfill on Carbonate Rock

The Upper Ottawa Street landfill is located in the City of Hamilton, Ontario, on the upper edge of the Niagara Escarpment (Figure 3). The area surrounding the landfill has a thin veneer of glacial till that overlies a sequence of Paleozoic dolostone and limestone of the Lockport Formation and shale of the Rochester Formation (Figure 4). Much of the waste was disposed of in quarries in the bedrock. Groundwater flow in this rock occurs in openings along nearly horizontal bedding planes and along nearly vertical joints. The intact rock has no significant permeability.

The landfill was operational from the early 1950's to 1980. It received normal municipal waste and industrial wastes of many kinds, including quantities of potentially hazardous liquid wastes. During the past two years the landfill surface has been contoured to promote surface runoff and a cover of flue dust and clay has been applied. Buried drains have been installed along much of the periphery of the landfill to pass leachate to a sewer connection.

Because the landfill was operated as a relatively uncontrolled dumping ground for a large industrial region and because of the quantities of potentially hazardous wastes that the landfill is known to have received, speculation regarding the fate of toxic chemicals that entered the landfill has been a major issue of public concern. In 1981,

various types of studies were initiated to determine whether or not the landfill represents a hazard to public health and to the environment. One of these studies is an on-going hydrogeological investigation for which the multilevel monitoring device described in this paper was initially designed and applied.

To date, twenty multilevel devices have been installed in the vicinity of the Upper Ottawa Street landfill. They were installed in 7.5 cm diameter boreholes to depths ranging from 23 to 46 m. The holes were drilled by a track-vehicle mounted rig (CME75) using hollow-stem augers to penetrate the glacial till above the bedrock. Continuous cores of the bedrock were then taken using a conventional core barrel with a water-circulation diamond bit. The cores were used to identify the fracture network and to select zones of large or frequent fractures that would be suitable for groundwater monitoring. The depths of these zones were often confirmed by water-loss information provided by the driller. The length of the sampling intervals ranged from 0.6 m to 3.0 m, however most were 1.3 m. The locations of the monitoring zones at nine of the multilevel sites are shown in Figures 4 and 5.

The majority of the multilevel devices contain five piezometers and the remaining devices have six piezometers. Most have single short packers placed at the top and bottom of each piezometer interval but some were installed with double short packers at the top and bottom of each interval. In many cases a single packer was placed in the blank interval between monitoring zones. The packers used in the last three devices had double strap clamps and silicone sealant in the clamp grooves to prevent leakage of water along the rubber sleeve. The

earlier devices had only single strap clamps. The majority of the devices were installed without the drill rig at the borehole. No major problems were encountered during the installation of the multilevel devices. A temporary blockage did occur in one hole however, apparently caused by rock rubble blown by methane gas into the borehole from the borehole wall.

Dye was injected into the casings of most of the multilevel devices to identify monitoring zones affected by leakage from the casing. This was of particular concern with regard to the devices installed earliest at the site before design modifications were made to minimize the potential for leakage. Dye was observed in some of the piezometers in the devices installed in the early stages of the study. In many cases very weak colour intensity of the water indicated only slight leakage. It is likely that the leakage was occurring at the elbow connections to the casing and so changes were made in later versions of the device to ensure a tight connection. The presence of the dye in a number of the piezometers led to experimentation with bentonite slurry as a means of sealing the interior of the casings. Based on this experimentation, the use of bentonite slurry as a method for finalizing the multilevel installations became a routine part of the complete monitoring device.

The primary purpose of the multilevel devices installed at the Upper Ottawa Street landfill site is to obtain vertical profiles of hydraulic head and water chemistry. A secondary purpose is to obtain, using techniques described by Hvorslev (1951), hydraulic-conductivity values from rising or falling-head tests of the monitoring zones.

Each of the multilevel devices at the site exhibits equilibrium vertical water-level differentials of considerable magnitude. The maximum differential in each device, which is the difference between the deepest equilibrium water-level and shallowest water-level elevation in the piezometers ranges from about 1 m to more than 20 m. Most maximum head differentials are between 3 and 10 m. Considering that the depth of the multilevel devices is less than 50 m, the vertical differentials are quite large.

It is reasonable to expect that the dominant flow of the groundwater in the shallow bedrock at most locations is nearly horizontal. This expectation is based on the fact that a considerable number of horizontal or nearly-horizontal fractures were detected in each borehole and the fact that the major lithologic units (i.e. shale, dolomite, cherty dolomite, shaley dolomite) form a nearly horizontal layered sequence of greater or lesser permeability zones. Inclined boreholes have not yet been drilled at the site and therefore not much is known about the vertical hydraulic conductivity.

The vertical profiles of hydraulic head obtained from the multilevel devices provide an indication of the tendencies for upward or downward flow at each monitoring site. Four examples of hydraulic-head profiles are provided in Figure 5. At many of the twenty monitoring sites, the head differential between the shallowest piezometer and the next shallowest piezometer is small or non-detectable. This condition is exemplified by Figures 5b and 5d, which apparently reflects the very fractured condition in the uppermost part of the bedrock.

At nearly all of the monitoring sites the hydraulic-head levels are lowest in the deepest piezometer or the next to deepest piezometer and are highest in the shallowest or next to shallowest piezometers. Examples of these conditions are provided in Figures 5a, 5b and 5c. These head differentials indicate that there is therefore a tendency for downward leakage of groundwater from the shallow zone. At some of these monitoring sites the deepest piezometer has a head level deeper than any of the other piezometers (Figures 5b and 5c). At these sites there is a possibility that some of the contaminated water emanating from the landfill is migrating to depths below the deepest zone in which piezometers currently exist.

At five of the monitoring sites the hydraulic head in the deepest piezometer is above the head in the shallowest piezometer (Figure 5a) and therefore at these sites there is upward leakage of water from deeper zones in the stratigraphic sequence. As more detailed data on the stratigraphy, fracture frequency and hydraulic conductivity are acquired from the site, it will be possible to evaluate the causes of the head differentials observed in the multilevel monitoring devices.

Nearly all of the piezometers in the multilevel devices have been sampled for analysis of chloride (Cl^-) and total dissolved organic carbon (TOC). These parameters are analysed quickly and at low cost and they provide a preliminary means of identifying the chemical status of the water. The multilevel devices that were installed early in 1982 have been sampled many times and those that were installed later have been sampled only a few times.

The initial task in the water-quality characterization program is the removal of drilling water from the monitoring zones. Progress with respect to drilling water removal is tracked semi-quantitatively by repeated samplings until the Cl^- and TOC analyses attain values that appear to be representative of formation water. The water used for drilling was obtained from the City of Hamilton water-supply system. It is very low in TOC and Cl^- . Both the natural groundwater and landfill-contaminated groundwater are much higher in these constituents.

Most of the piezometers are in zones that are not sufficiently permeable for continuous pumping to occur. In these cases, the standing water column is pumped out, the water level is allowed to recover and then the column is pumped out again. Because the rock in many places is only slightly or moderately permeable, recovery often takes many hours or even a few days. In general it was necessary to remove the standing volume of water many times before the Cl^- and TOC results indicated that the piezometer was primarily yielding formation water. It is difficult to determine whether or not a piezometer is still yielding a small percentage of drill water, except in those piezometers where the Cl^- and TOC values are consistently exceptionally high.

The results of the preliminary sampling program indicate that nearly all of the piezometers yield water that has concentrations of Cl^- and TOC, one or both of which are moderately or extremely high (Figure 5). Cl^- values generally range from several hundred milligrams per litre to several tens of thousands of milligrams per litre. Values of TOC range from several tens of milligrams per litre to several thousand milligrams per litre. Cl^- and TOC differences of more than a factor of

10, for each of these parameters, commonly occur within each multilevel device.

In investigations of groundwater contamination at landfills, values of Cl^- and TOC that are considerably above a few tens of milligrams per litre are normally strongly indicative of leachate contamination. At the Hamilton site however, these parameters are not conclusive indicators of leachate because uncontaminated groundwater in the shale and shaley dolomite can also have high levels of these constituents derived from the rock. Detailed investigations of both the inorganic and organic chemistry of groundwater at this site are in progress.

An Abandoned Landfill on Shale

The Bayview Park landfill is located in the City of Burlington, Ontario and occupies approximately 18 hectares (Figure 3). It was operational between the early 1960's and early 1970's, during which time it received largely municipal waste and some industrial waste from the City of Burlington. Like the Hamilton landfill, the Burlington landfill is situated on the edge of the Niagara Escarpment where bedrock is either exposed at the surface or covered by a thin veneer of clay till. The bedrock is red Paleozoic shale of the Queenston Formation, which is essentially continuous to depths of over 100 m except for occasional green shale beds which rarely exceed 0.2 m in thickness. The top few meters of the shale are extremely weathered. Below this, the shale has an extensive fracture network that appears to have developed mainly along horizontal bedding planes. Groundwater movement is towards the southeast along the slope of the escarpment and occurs primarily through this network of small fracture openings.

Between 1979 and 1981 investigations were conducted at the site by hydrogeological consultants (Morrison Beatty Limited, 1981), who reported that contaminants from the refuse have been introduced to the groundwater moving in the shale. In the spring of 1982, an investigation was begun to determine the extent of the zone of contamination and the nature and influence of attenuation processes on contaminant concentrations in this zone. Six multilevel devices were installed in the vicinity of the landfill to compliment the existing network of conventional piezometer nests (Figures 6 and 7). At locations WAT 1 and WAT 2, devices were installed to depths of 30 m and 46 m, respectively. The remaining devices, at locations WAT 3, 4, 5 and 7, were all installed to a depth of 38 m.

The first multilevel device (WAT 1) installed at the site was completed using the same methods described above for the Hamilton site. A 7.5 cm diameter borehole was cored to determine the thickness of the refuse and to examine the stratigraphic sequence and fracture network in the shale. Five sampling intervals, each about 0.6 m in length, were isolated by single packers. The 9 mm ID polyethylene piezometer tubes were contained in a 5 cm ID Schedule 40 PVC casing and the elbows were sealed by a threaded nylon nut tightened on an O-ring. Installation of the device was done in 6.5 m sections using the drill rig winch. Lifting these long sections put considerable stress on the 90° elbows and caused some of the coupling seals to crack.

During the next phase of the drilling program, an air-tricone method of rotary drilling without coring was used. This change allowed the drilling to proceed more quickly without the use of significant

amounts of drill water. To maximize the possibility of intersecting fractures along which contaminant migration may be occurring, the borehole diameter was increased to 11 cm. The inside diameter of the PVC casing of the multilevel device was then increased to 7.5 cm in order to accommodate as many as 12 piezometer tubes.

The polyethylene piezometer tubes in these larger devices were increased from 9 mm ID to 12 mm ID to allow more clearance for sampling equipment and for water-level tapes. In addition, the piezometer tubes were taped together at 0.7 m intervals along their entire length to eliminate spiralling or twisting within the PVC casing. An open-ended piece of 9 mm tubing about 5 m in length was also added to the multilevel device to facilitate water level measurements in the casing. Twelve 12 mm tubes is the maximum number that can be used with a 7.5 cm ID PVC casing.

Boreholes at WAT 2, WAT 3 and WAT 4 were drilled with the air tricone method using a 10.8 cm (4.25 inch) bit. Hollow stem augers (10 cm ID and 17.5 cm OD) with centre rods were used to drill through the thin overburden and into the top few meters of extremely weathered shale. The augers were left in place to prevent caving of the hole and escape of air at shallow depths while tricone drilling proceeded through the hollow stem. Water flowing from the formation into the borehole produced a wet slurry of rock cuttings that was easily brought to the surface from as deep as 46 m using a large capacity air compressor.

During this phase of the drilling, the casing of the multilevel devices were assembled in ten 3.3 m sections to facilitate handling during installation and to prevent the stress on elbows and couplings

encountered earlier in the field program. Each section consisted of a 1.3 m sampling interval with 0.25 m long packers at the top and bottom, and a 1 m length used as a spacer between sampling intervals. Glass wool was taped over all sampling ports to act as a filter for the high sediment content of the water.

A major problem became apparent when these devices were installed. The outside diameter of couplings for standard 7.5 cm Schedule 40 PVC pipe varies slightly, and in many cases it was only slightly smaller than the actual borehole diameter. This causes a tight fit in the borehole, which on several occasions necessitated the use of the drill rig to push the device into place. During the next phase of the drilling program, boreholes were drilled with a 11.3 cm (4.5 inch) tricone bit which successfully eliminated the use of strong downward force on the devices during installation.

The difficulty of achieving a totally non-leaky installation increases from the six-piezometer devices used at the Hamilton site to the twelve piezometer devices used at the Burlington site. Fluoranine dye injected into the casing of each device identified some leakage in the three multilevel devices installed earliest at the site. The construction and installation modifications made during the later stages of field investigation appear to have minimized the occurrence of dye to a maximum of two sampling intervals per device. It is expected that additional design modifications will further minimize the occurrence of dye in future installations.

The multilevel devices installed at the Burlington site have thus far provided equilibrium water levels and a few sets of Cl^- and TOC

analyses. Figure 8 shows the sampling intervals and vertical profiles of hydraulic head, Cl^- and TOC for four multilevel devices.

The hydraulic head profiles for all of the multilevel devices exhibit major features that would generally not have been detected in a conventional piezometer nest having only a few piezometers. Head differentials at WAT 2 are downward at depths shallower than about 30 m and upward at greater depths. In WAT 1, situated near the downgradient edge of the landfill (Figures 6 and 7), the piezometer ports are all deeper than 30 m below the landfill surface and have upward hydraulic-head differentials.

The results obtained from the other four multilevel devices, all of which are located downgradient of the landfill, are shown in Figure 7. Each of these multilevel devices exhibits large hydraulic head differentials. At WAT 3 the head profile is irregular and has no distinct upward or downward head differential trend. At WAT 4 there is a gradual decline in head with depth. At WAT 5 and 7 the head values in the lower piezometers are much deeper than those in the shallow piezometers. At WAT 5, the largest decline in head occurs about one-third of the way down the profile, whereas at WAT 7, a large abrupt decline occurs two-thirds of the way down the profile. The characteristics of the head profiles seen in all of the multilevel devices are probably related to the bedding features and associated fracture permeability in the shale. These relationships are currently being investigated.

Background water chemistry at the site is monitored at WAT 2, located upgradient of the landfill (Figure 6). Here, Cl^- and TOC

concentrations increase markedly with depth from minimum values of 100 and 400 mg/L at the water table to values about one order of magnitude larger at the bottom of the multilevel device 48 m below. The vertical profiles of Cl^- and TOC at WAT 3, 4, 5 and 7 show large variations in concentration within each device as well as distinct differences between multilevel devices.

The large number of available piezometers in each multilevel device provides many sampling options. Initially, all of the piezometers in each device need to be sampled and, depending on the initial results, a sampling strategy can be developed. At the Bayview site, all of the monitoring points were sampled to determine the vertical distributions of Cl^- and TOC. These profiles can now be used as a guide to select piezometers for more specific organic sampling. The presence of organic compounds in the groundwater will be more diagnostic of landfill-derived contamination than the Cl^- and TOC concentrations which exhibit very large natural fluctuations in the bedrock.

A Dam Site At A Tailings Impoundment

Multilevel devices are being used in a third field investigation, which is taking place at a dam site in the Elliot Lake uranium-mining district of Northern Ontario. The devices were installed in late 1982 to determine the groundwater conditions as part of a geotechnical assessment of the dam foundation.

The foundation is located within a saddle of an east-west trending bedrock ridge that separates a tailings basin from an effluent treatment pond (Figure 9). The water-level elevation difference between these two

ponds is approximately 11 m. Virtually no overburden occurs within the saddle area. A spruce bog up to 10 m deep infills a bedrock depression located immediately south of the centre line of the dam proposed.

The bedrock underlying the site is comprised of metamorphosed interbedded arkose, arkosic greywacke, fluvial pebble-cobble conglomerate and conglomeratic greywacke (tillite) as shown in Figure 10. A vertical diabase dyke, 18 to 25 m in width, occurs beneath the upstream shell of the dam. The sedimentary strata were subjected to low grade regional metamorphism and they were gently folded during early Proterozoic time. The silicious composition of the sedimentary rocks combined with the effects of this regional metamorphism have resulted in a highly indurated rock mass with virtually no intergranular porosity. Subsequent fracturing of the rock and related weathering along these discontinuities has resulted in variable degrees of fracture-controlled hydraulic conductivity.

Subsurface investigations of the bedrock were carried out by coring four NQ size (7.5 cm) boreholes. The locations of the boreholes, numbered 601 to 604, are shown on Figure 9. They range in length from 32 to 45 m. One of the holes is vertical and the other 3 are inclined 60° from horizontal. The holes were advanced in 3 m intervals. The permeability of the bedrock was determined by falling-head injection tests conducted in the bottom of the borehole after each 3 m interval of drilling. Between 9 and 13 tests were done in each borehole.

Multilevel devices containing between 4 and 6 piezometers were manually installed in each completed borehole. Specific borehole intervals for piezometer ports were selected based on a detailed

evaluation of results of the falling-head hydraulic conductivity tests and of the core, which included a rock quality designation (R.Q.D.). A summary of these types of data is provided in Figure 3 for Borehole 602.

The piezometer intervals were isolated at the top and bottom by a double set of 20 cm long packers. Initially, water poured into the casing leaked through the packers because the rubber packer sleeves were fastened by only single strap clamps. However, swelling of the packers sealed these leaks within two days of installation. In this time the water level in the casing stabilized, indicating that there is no hydraulic communication between the interior of the piezometer casing and the boreholes.

The water levels recorded in the device installed in Borehole 602 are provided as an example of the data obtained at this study site. A very large (10 m) hydraulic head differential (Figure 11) exists between piezometers 4 and 5 in this device. Large head differentials were also observed in the other three multilevel devices.

Water-level monitoring indicates that the shallowest water level in each multilevel device is in the range of 1 to 5 m. These water levels are probably representative of water-table conditions within the rock. The deeper piezometers have much deeper water levels, which range up to 14 m below surface. These levels indicate strong downward hydraulic gradients within the bedrock ridge. At borehole 603, where the water table is at 4 m below ground surface, the water levels in the deeper piezometers are nearly at the level of the piezometer ports, indicating downward hydraulic gradients that have a magnitude near unity.

The results of water-level monitoring at this site have demonstrated that the multilevel device is capable of indicating equilibrium water levels in situations where extremely large head differentials exist. The large differentials demonstrated by the devices also provide an indication that in rock with sparse fractures, the double short packers are capable of isolating the monitoring zones.

As part of the investigation, a pumping test was conducted to determine the degree of hydraulic connectivity between the various fractures in the bedrock. Borehole 604, which encountered a significant zone of permeable fracturing adjacent to the diabase dyke, was pumped using an air compressor prior to the installation of a multilevel device in the hole. Multilevel devices installed in Boreholes 601, 602 and 603 were monitored over a 24-hour period to identify any response to pumping. Water-level changes in Borehole 603 indicated that two fractures occurring in piezometer intervals 1 and 2 are hydraulically connected to the pumped zone. No response was recorded in any of the other piezometers at the site.

Groundwater samples from the multilevel devices were collected two months after installation. A small polyethylene tube, attached at the surface to a gas regulator was inserted to the bottom of each piezometer tube. Nitrogen gas was then used to expulse the standing water column in a steady stream. A waiting period of 1 to 5 minutes was required between such flushings to allow recovery of the water column. This method was used to flush several litres of water from each piezometer. During the flushing, the electrical conductance of the groundwater was monitored. Samples were collected for chemical analysis when the

electrical conductance stabilized. A summary of the conductivity, pH and alkalinity of water samples from piezometers in Borehole 602 is shown on Figure 11.

At each monitoring site each of these parameters increase with depth. At Borehole 602 there is a sharp increase in pH and alkalinity from piezometer 2 to piezometer 3 (Figure 11). This suggests that these piezometers sample distinctly separate zones within the borehole. Piezometer 2 is in a 1.5 m zone containing a porous siltstone breccia of moderately high permeability (2×10^{-4} cm/s). Piezometer 3 is situated within a zone of low permeability (1×10^{-5} cm/s) in a massive arkose unit.

DISCUSSION

The multilevel monitoring devices installed at each of the three field sites show the existence of very large and in many cases abrupt hydraulic-head differentials between two or more of the piezometers in each hole. At most of the monitoring sites there are also large differences in water chemistry from one depth to another. Without monitoring at many depths at these sites, it would generally not have been possible to detect these large differences in head and chemistry.

At each of the field sites the local topography has considerable variability and therefore the local water table is also quite variable spatially. It is reasonable to expect that the large vertical differentials in head reflect spatial variations of the local water-table and large variability of the flow resistance along the various fractures.

The large differences in water chemistry from one depth to another probably reflect large differences in residence time within the various fractures and differences in the mineralogy of the fracture surfaces. The electrical conductance and chloride concentrations in the groundwater in the shale at the Burlington site and in the dolomite and shale at the Hamilton site are orders of magnitude higher than at the Elliot Lake site. The high values in the shale and dolomite are a result of the much greater availability of soluble salts in these rocks. The salts probably enter the groundwater in the fractures by molecular

diffusion from the slightly porous rock matrix. At some of the monitoring sites, landfill leachate also contributes to the high salt concentrations.

The large differences in hydraulic head and water chemistry exhibited by the multilevel devices in each of the three field areas suggests that the packers are effective borehole seals. Whether or not any of the packers allow any significant vertical leakage along the borehole from one monitoring interval to another cannot be conclusively deduced from the hydraulic head and water-chemistry data. In this sense the dilemma associated with the possibilities for seal leakage has some similarity to that associated with a conventional standpipe piezometer in a single borehole in which a short bentonite or grout seal above the sandpack isolates the sandpack from the backfilled borehole above.

The multilevel device however, offers a possibility for detecting leaky seals. Water is pumped from one piezometer and the water levels in the piezometers above and below are monitored. If there is no change in these water levels, it can generally be concluded that the packers are providing an effective hydraulic seal. In cases where the piezometer ports occur in very permeable zones, it is possible that the precision of the water-level measurements would be inadequate to detect the effects of small leakage rates. In cases where the rock has fractures that provide vertical hydraulic connection from one sampling zone to the next, this pumping and monitoring method of evaluating packer seals would be inconclusive even if the packer forms a perfect seal in the borehole.

The probability that the packers will form a complete seal in the borehole can be enhanced if each packer interval is comprised of multiple packers or very long packers. With multiple packers or long packers, it is much less likely that grooves or channels on the borehole wall will extend unsealed along the entire packer interval. Laboratory observations have been made of the behaviour of the inflated packers in transparent plexiglass tubes. The inflation pressure exerted by the Dowell sealant gradually increases during the weeks following the initial expansion. It appears that the swelling pressure of the Dowell sealant is capable of forcing the rubber membrane into grooves and channels but the degree to which very small grooves or scratches on the borehole wall are sealed by the pressurized rubber membrane is not known. Longer packers or multiple packers provide greater assurance of achieving a total seal because small grooves and scratches are generally not very long.

In addition to or as an alternative to the use of more or longer Dowell seal packers, there exists another possibility for increasing the potential to achieve a complete hydraulic seal in boreholes that have rough or grooved walls. Rings of compressed swelling bentonite produced in high pressure molds (Desaulniers, 1983) can be placed around short blank lengths of casing that are coupled above and below the Dowell-seal packers.

The use of swelling bentonite rings as a means of providing a seal has been demonstrated in transparent plexiglass tubes in the laboratory but it has not yet been tried at a field site. Methods for reducing the fragility of the bentonite rings mounted on the casing segments need to be investigated.

One of the main differences between the piezometers in the multilevel device and most conventional standpipe piezometers or wells is the small diameter of the piezometer tubes. This provides a major disadvantage in some situations and advantages in other situations. The small diameter of the piezometer severely restricts the rate at which water can be pumped from the piezometers. As a result, the removal of drilling water from the borehole generally takes much longer than for conventional piezometers. This is a problem particularly in low-permeability zones because piezometers in these zones do not yield a continuous discharge of water when pumped. The storage volume in the narrow piezometers is so small that repetitive removal of storage volumes is a slow means of extracting the drilling water from the formation. The drilling water problem can be avoided if air-circulation drilling is used, but when continuous rock cores are desired, the use of drilling water is generally necessary.

Once the drilling water or other effects of drilling disturbance are removed, the piezometers in the multilevel device are well suited for groundwater-quality monitoring. Pumping of large quantities of water from the piezometers is not necessary and may even be undesirable. Pumping large water volumes can cause the distribution of water-quality parameters to be disturbed or mixed as water is drawn towards the piezometer from greater distances and from more fractures. If the sampling-induced disturbance is large, the monitoring results tend not to be reproducible or tend not to be representative of discrete zones in the formation. For fractured rock, little is known about the effects of increased pumping on sample reproducibility or on sample

representativeness. The pumping of large volumes of water for sampling, which is usually done when conventional larger-diameter piezometers are used for groundwater monitoring is not advantageous.

All of the multilevel devices that have been installed to date have been placed in relatively competent bedrock consisting of medium bedded dolostones, thin bedded shales, and arenaceous metasedimentary rocks. Consequently, caving of rock rubble in the boreholes did not occur, except in the single case where a large gas flow was encountered in a highly fractured zone. In rocks of a friable or incompetent nature such that boreholes may not stay open, considerable difficulties can be expected to occur when attempting to install this type of instrumentation. However, in such situations the construction of the device can be modified to decrease the overall diameter so that it can be installed through the drill rods following completion of the drilling. The device would have a smaller number of piezometers. Additional use of drill rig standby time would be incurred but under these rock conditions such problems would be unavoidable with any form of piezometer installation.

SUMMARY AND CONCLUSIONS

The multilevel monitoring device described in this paper provides a maximum of 12 narrow-diameter piezometers in 11 cm boreholes and 6 narrow-diameter piezometers in 7.5 cm holes. The use of the device at the three field sites has established that the device can be assembled quickly in the field and that it can be installed in open boreholes by a two or three person crew without the aid of a drill rig. Water levels in the piezometers are measured by means of simple electric probes and water samples are acquired by vacuum line if the water level is less than about 8 m, or by gas-lift or gas-drive sampling if the water levels are deeper. For normal investigations of groundwater quality, the piezometers are constructed of polyethylene or polypropylene tubing and PVC casing. For monitoring at sites where trace levels of specific organic compounds are of primary interest, it may be desirable to use Teflon or Teflon-lined piezometer tubes and stainless steel casing in the monitoring intervals.

The monitoring device has several advantages over conventional piezometers and observation wells. A primary advantage is that only one borehole is necessary to provide space for many sealed piezometers, whereas nests of conventional piezometers require many boreholes. Other advantages include the relatively rapid response times for the narrow-diameter piezometers, and the lack of bentonite or grout seals directly in contact with the sampling zone.s. Bentonite or grout seals may

influence the chemical composition of water samples drawn from low-permeability zones where the water-contact time is long. When bentonite or grout is used to form seals for conventional piezometers in zones where groundwater is saline or high in some types of organic compounds, the seals may undergo reactions that cause the integrity of the seals to decline. When the packed off intervals between monitoring intervals are small, the multilevel monitoring device makes it possible to obtain water samples from segments in a single borehole that, in total, cover essentially the entire borehole. The possibility of not encountering zones of contamination at sites where contamination exists only in thin depth zones is therefore minimal. It is also possible to isolate short monitoring intervals so that individual fractures or narrow fracture zones can be monitored. The device is suitable for use in angle holes as well as vertical holes.

The major disadvantage of the multilevel device relative to a conventional large-diameter standpipe piezometer is that the rate at which water can be pumped from the piezometers is very small. It is adequate for nearly all types of chemical analyses but may become restrictive when it is necessary to pump out large volumes of water to remove the effects of drilling water. Slow pumping is particularly tedious in diamond-drilled core holes where the use of drill water is difficult to minimize.

The recent experience acquired at three field sites in different types of rock indicates that considerable design precautions and care during installation are necessary to avoid leakage from the casing into the borehole. The casing water contains chemicals dissolved from the

Dowell sealant and therefore leakage introduces uncertainty when the device is used for groundwater-quality monitoring. The potential for leakage to occur has been considerably reduced by the recent improvements in the manner in which the rubber packer sleeve is secured to the casing. Even if leakage is detected by water-level monitoring in the casing, adverse long-term effects can be minimized after inflation of the packers by replacing the water column with a bentonite slurry.

Monitoring and sampling of the multilevel devices at the three field sites showed many large vertical differentials in hydraulic head from one piezometer to the next. The hydraulic conductivity commonly varied considerably from one depth to another. When pumped, some piezometers yielded water continuously whereas others would only yield water discontinuously at very slow rates. Large variations in water chemistry with depth are common. The results are consistent with the premise that the complexity and variability in the hydraulic and hydrochemical conditions within fracture systems warrants the acquisition of data from many levels at each monitoring site. The multilevel device provides a relatively inexpensive means of obtaining such data.

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investigation that there is a need for research and development pertaining to methods for groundwater monitoring in fractured rock. These individuals provided the necessary administrative support that initiated the development of the device.

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FIGURE CAPTIONS

Figure 1. Schematic illustration of the multilevel monitoring device (a) discrete-interval monitoring and (b) segmented-borehole monitoring.

Figure 2. Schematic illustration of the components of the multilevel device (a) the packer system and (b) the monitoring port.

Figure 3. Location of the Hamilton and Burlington study sites.

Figure 4. Stratigraphy and isohydraulic head lines along an east-west cross section at the Hamilton site.

Figure 5. Depth profiles of hydraulic head, chloride and total dissolved organic carbon obtained from four multilevel devices at the Hamilton site.

Figure 6. Map showing surface drainage features and locations of multilevel sampling devices at the Burlington site.

Figure 7. Hydrogeologic cross section along the main direction of groundwater flow at the Burlington site.

Figure 8. Depth profiles of hydraulic head, chloride and total dissolved organic carbon obtained from four multilevel devices at the Burlington site.

Figure 9. Physiographic features and locations of multilevel devices at the tailings impoundment site.

Figure 10. Hydrogeologic cross sections at the tailings impoundment site.

Figure 11. Depth profiles of hydrogeologic parameters at borehole 602 at the tailings impoundment site.

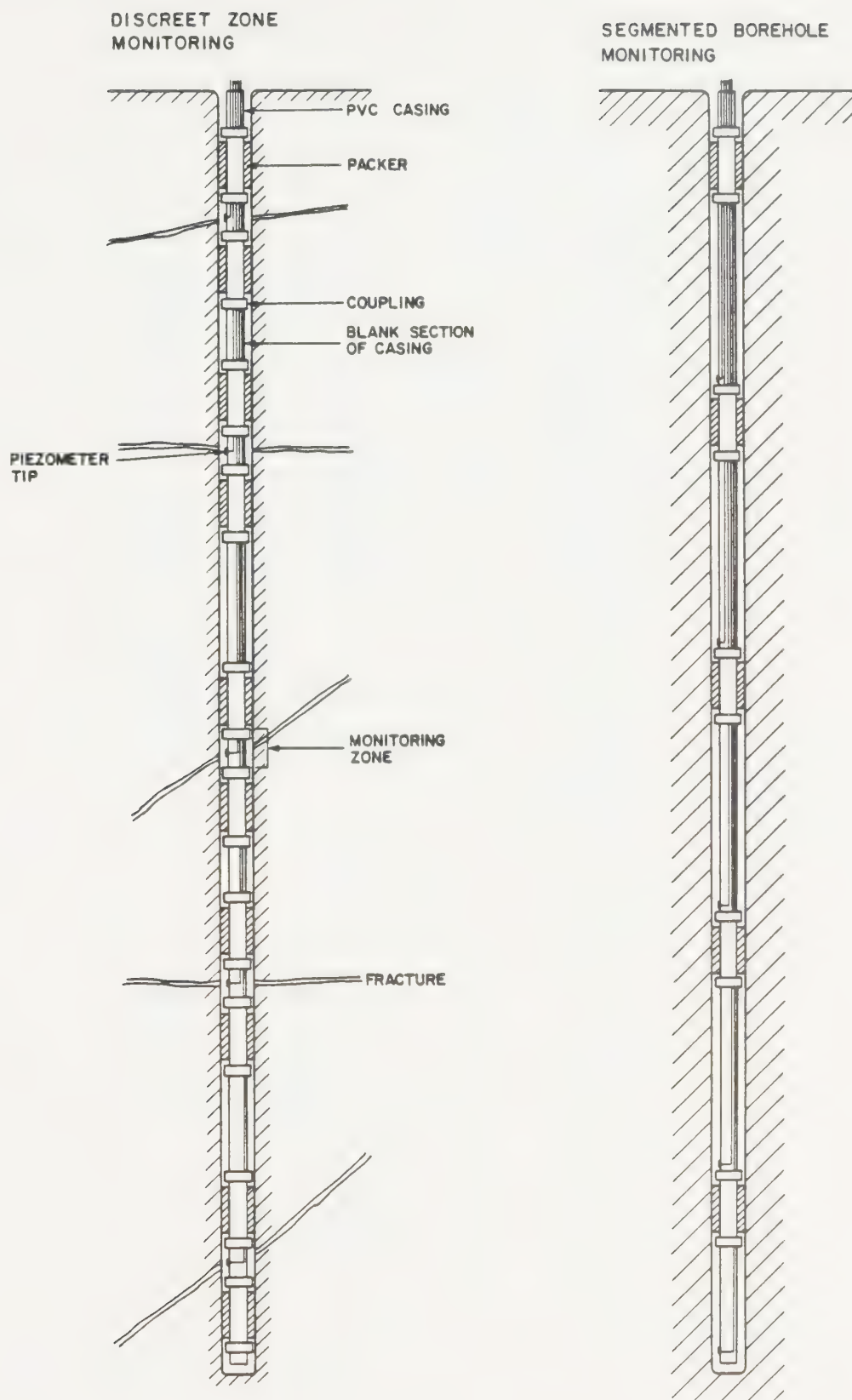
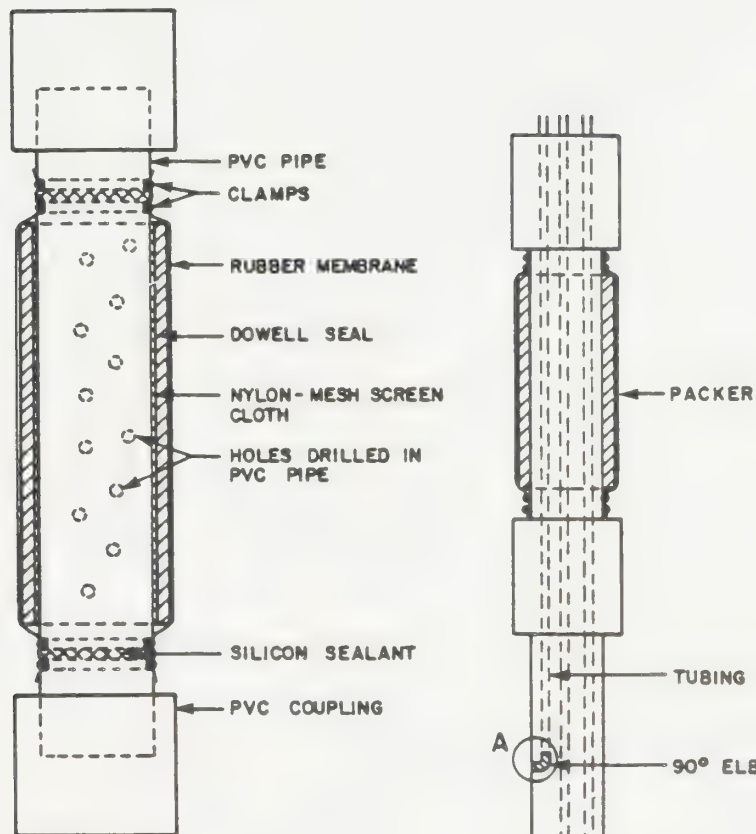


Figure 1. Schematic illustration of the multilevel monitoring device (a) discrete-interval monitoring and (b) segmented-borehole monitoring.

DETAIL PACKER



DETAIL A

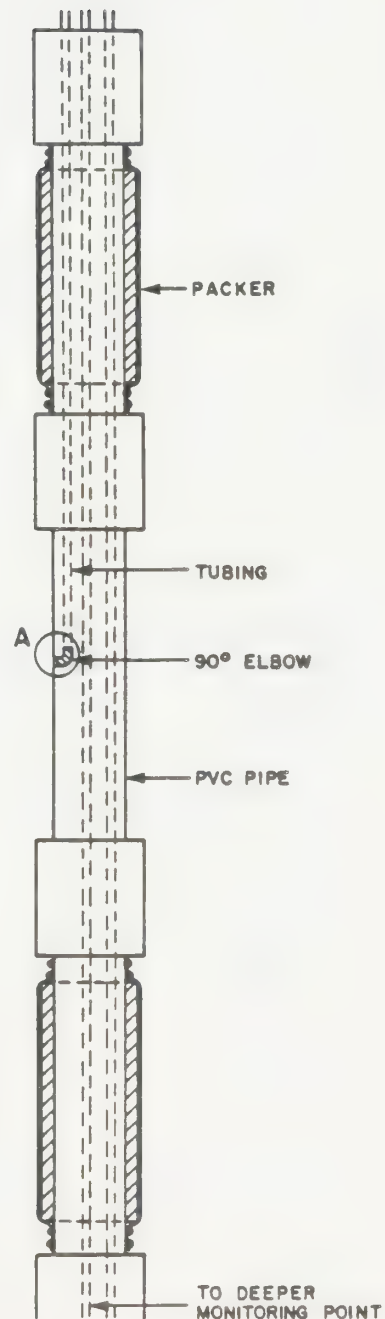
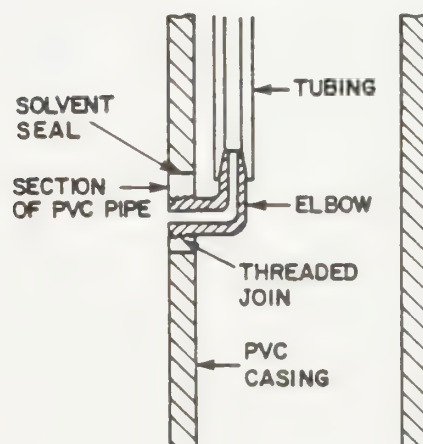


Figure 2. Schematic illustration of the components of the multilevel device (a) the packer system and (b) the monitoring port.

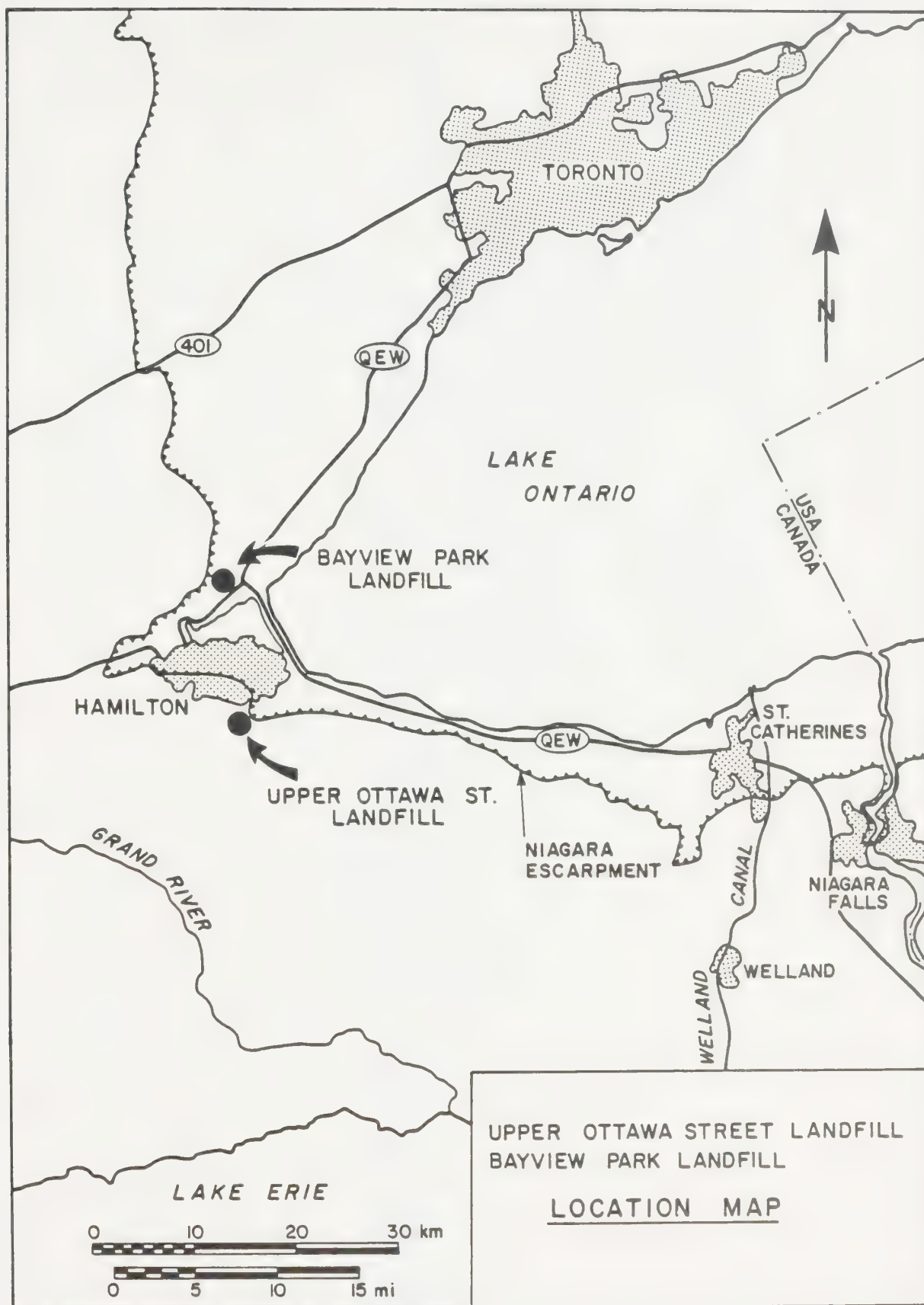


Figure 3. Location of the Hamilton and Burlington study sites.

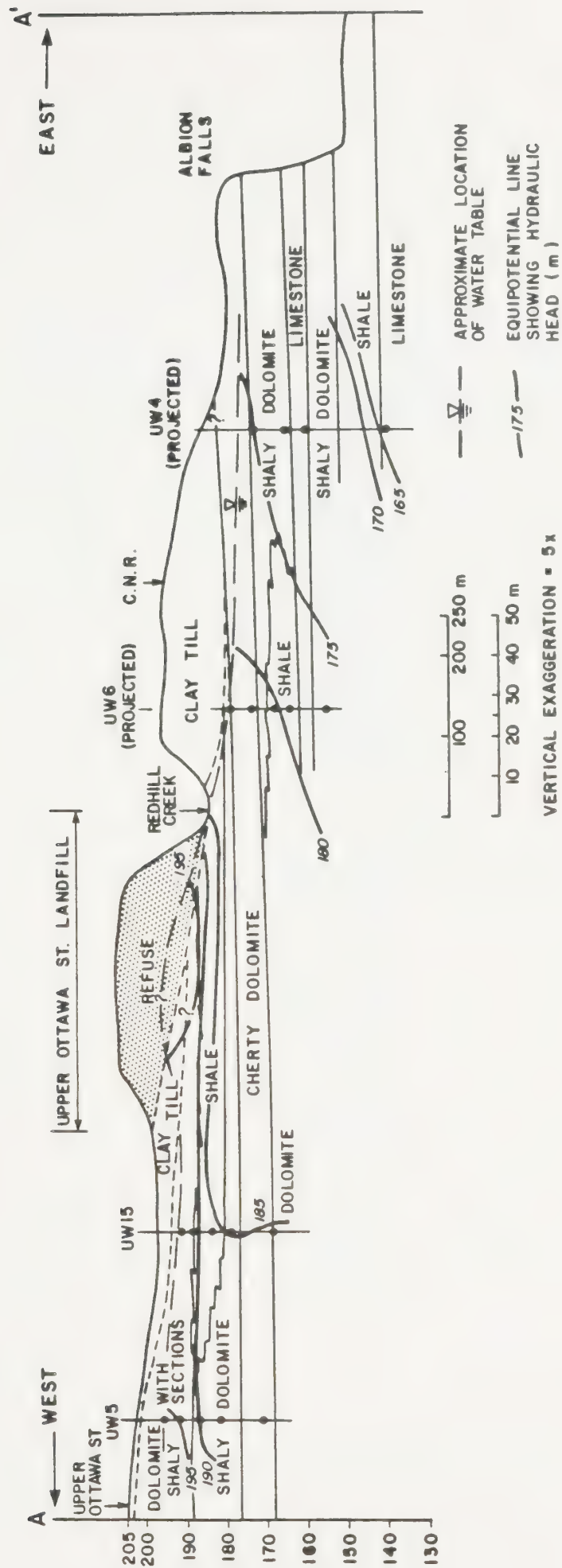


Figure 4. Stratigraphy and isohydraulic head lines along an east-west cross section at the Hamilton site.

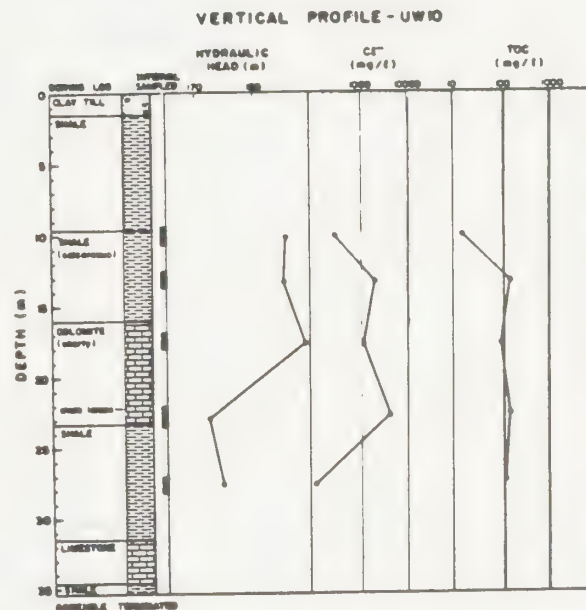
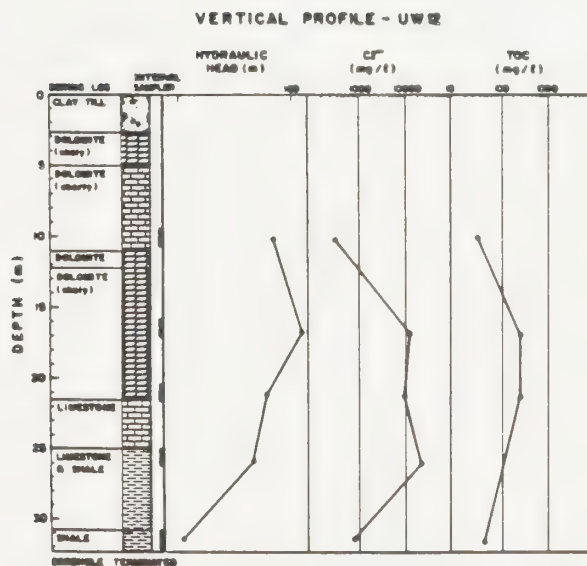
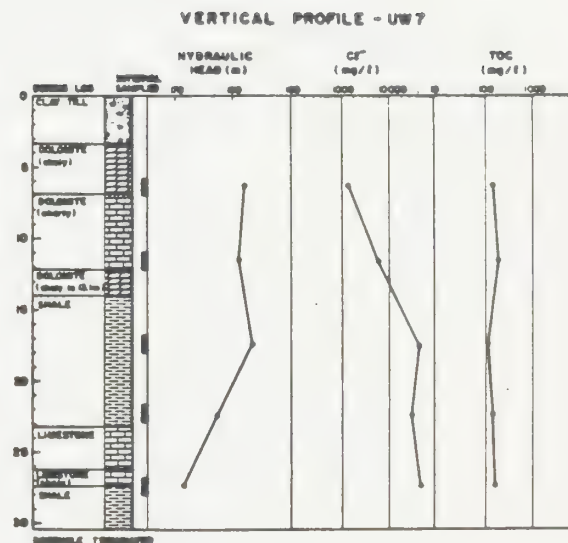
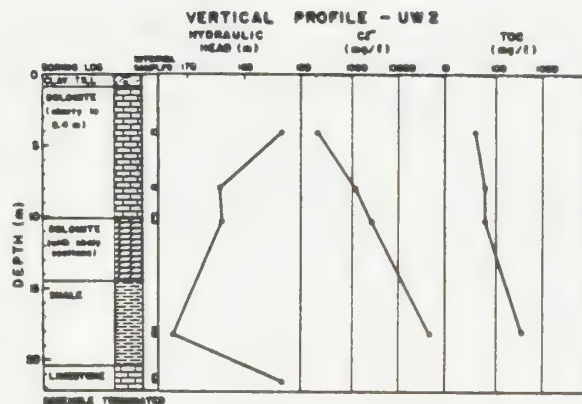


Figure 5. Depth profiles of hydraulic head, chloride and total dissolved organic carbon obtained from four multilevel devices at the Hamilton site.

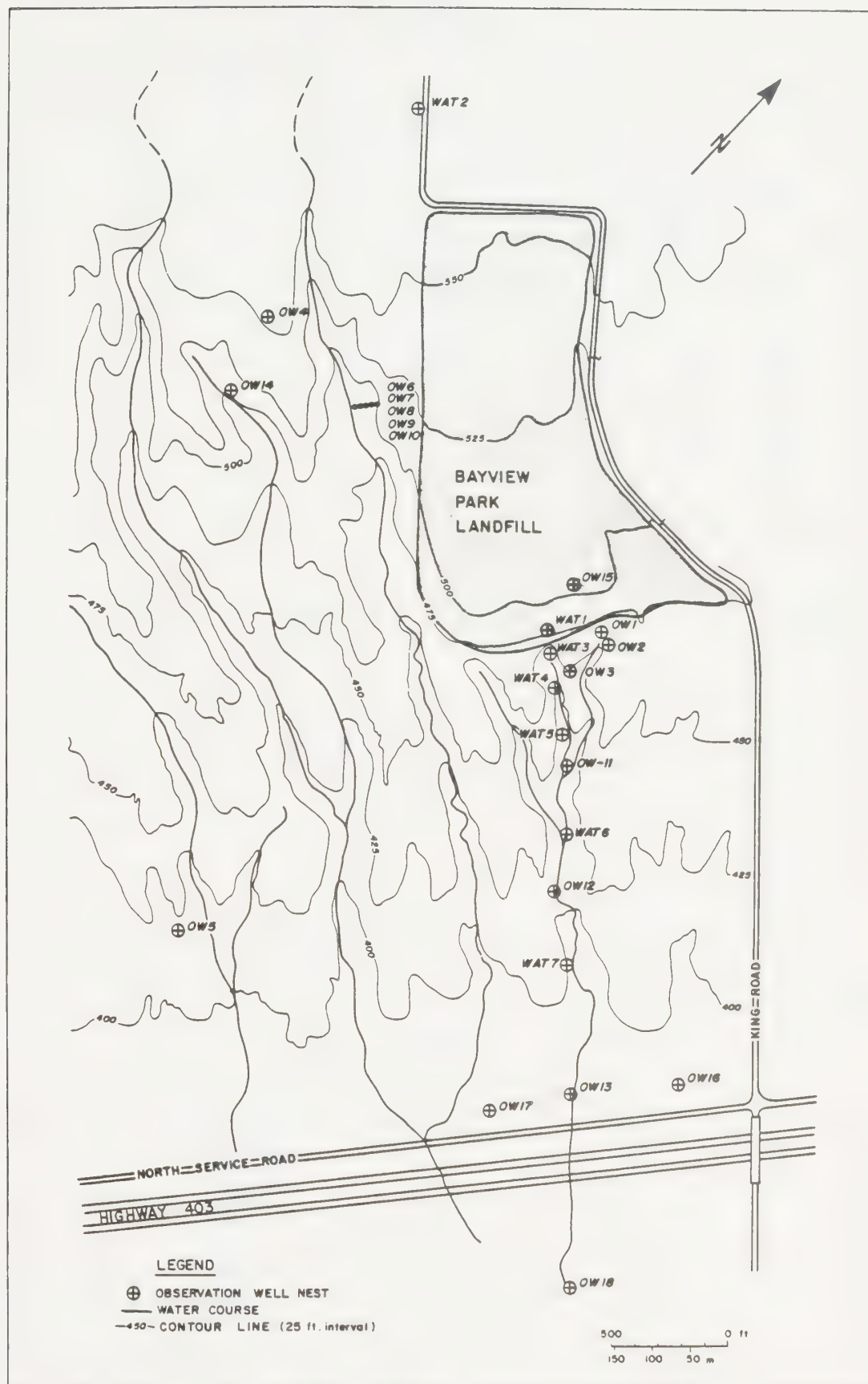


Figure 6. Map showing surface drainage features and locations of multilevel sampling devices at the Burlington site.

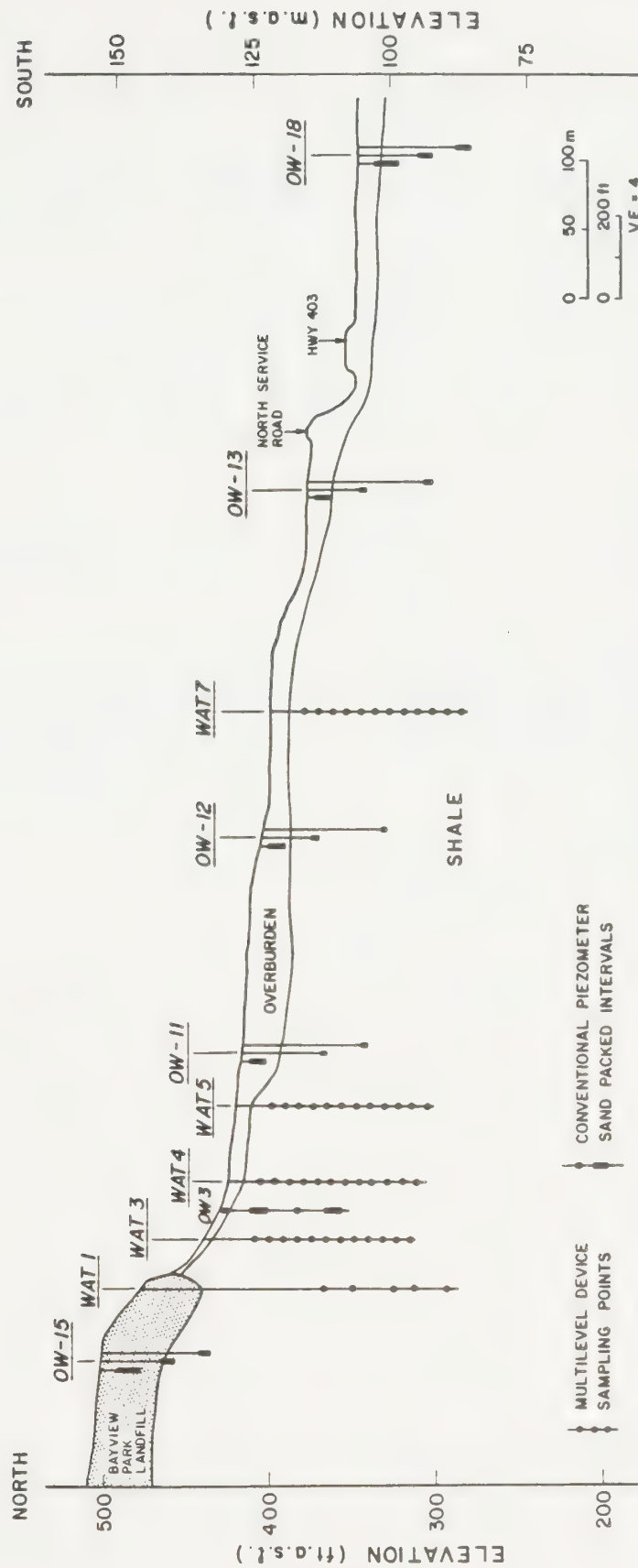


Figure 7. Hydrogeologic cross section along the main direction of groundwater flow at the Burlington site.

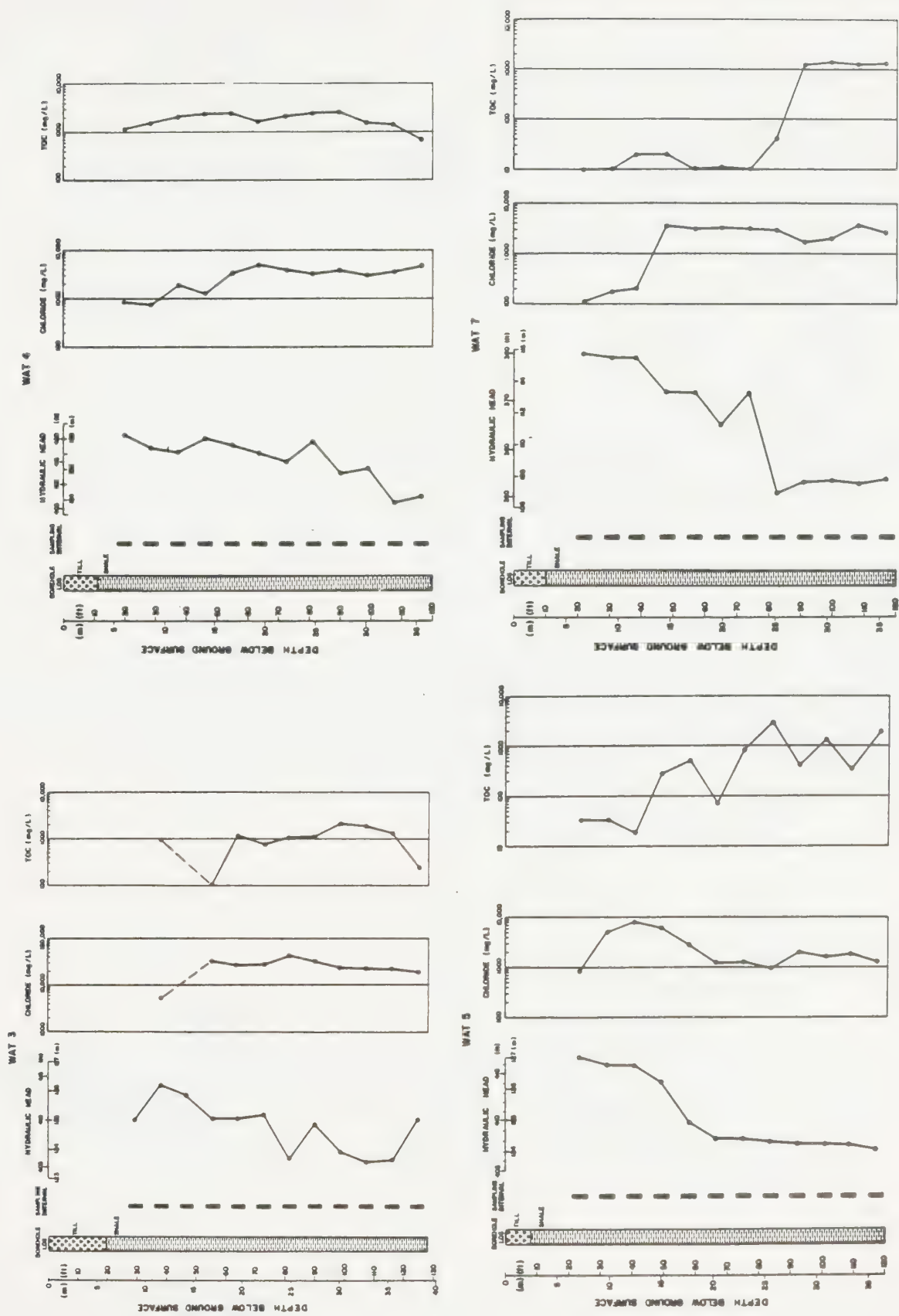


Figure 8. Depth profiles of hydraulic head, chloride and total dissolved organic carbon obtained from four multilevel devices at the Burlington site.

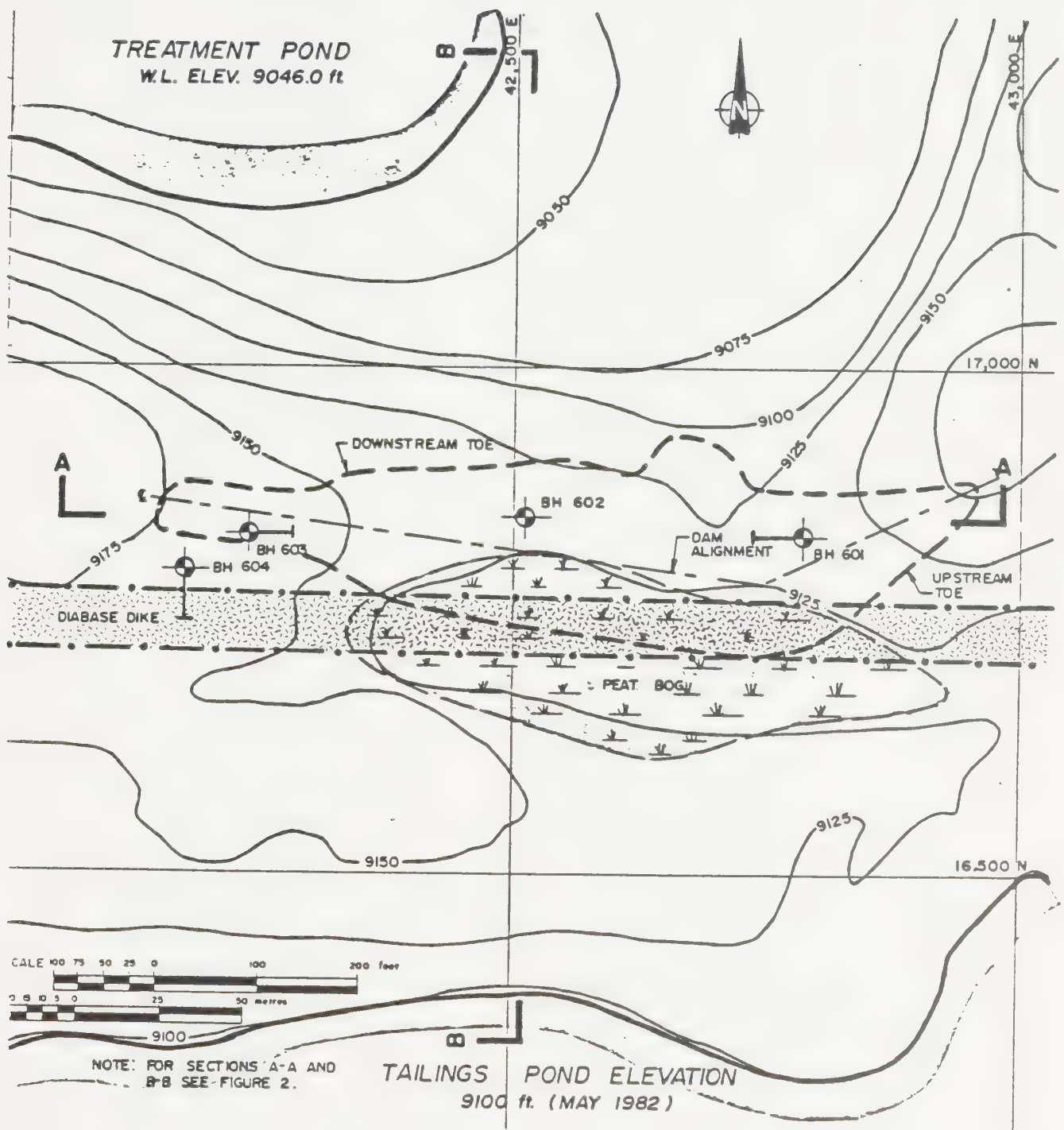
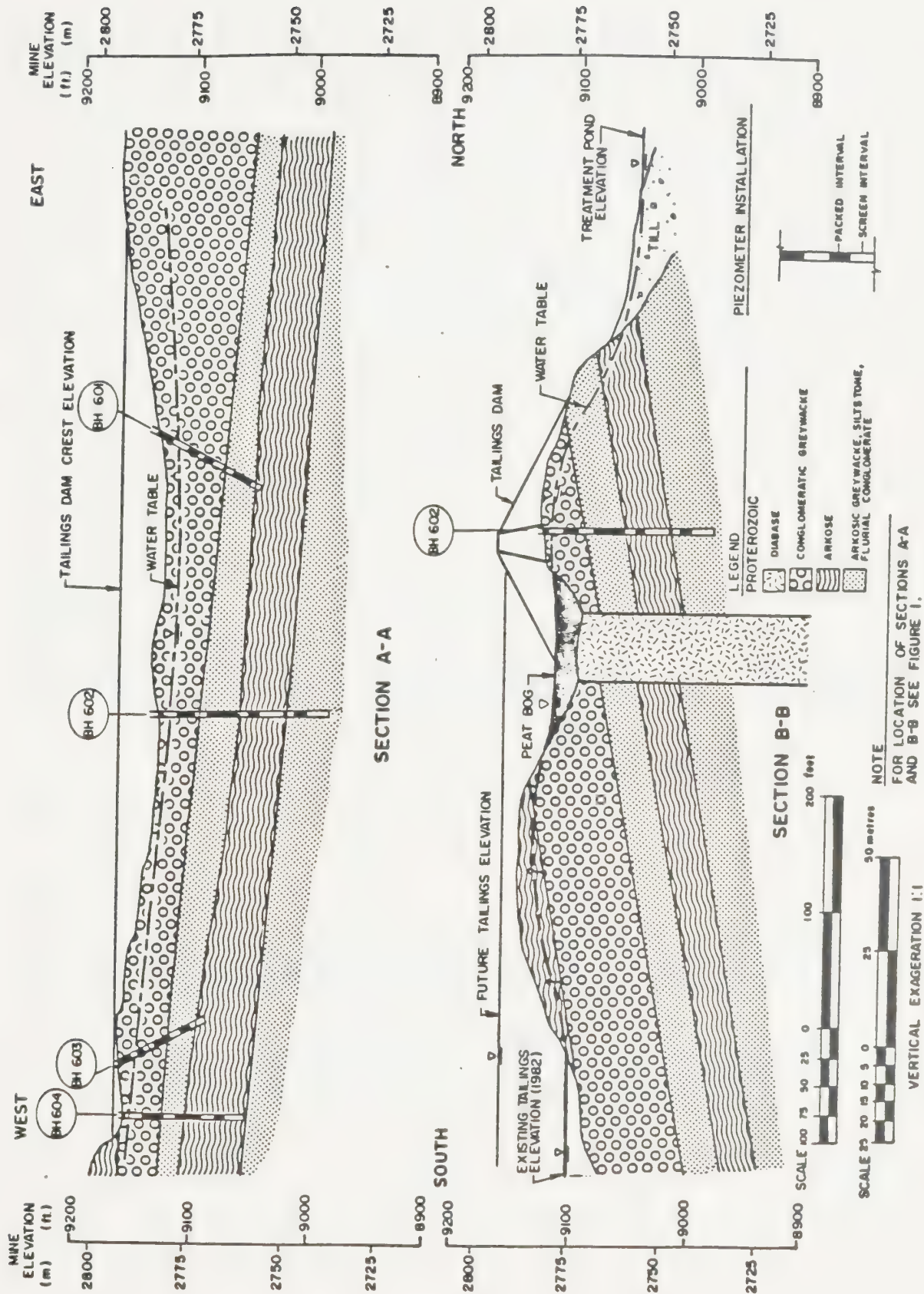


Figure 9. Physiographic features and locations of multilevel devices at the tailings impoundment site.



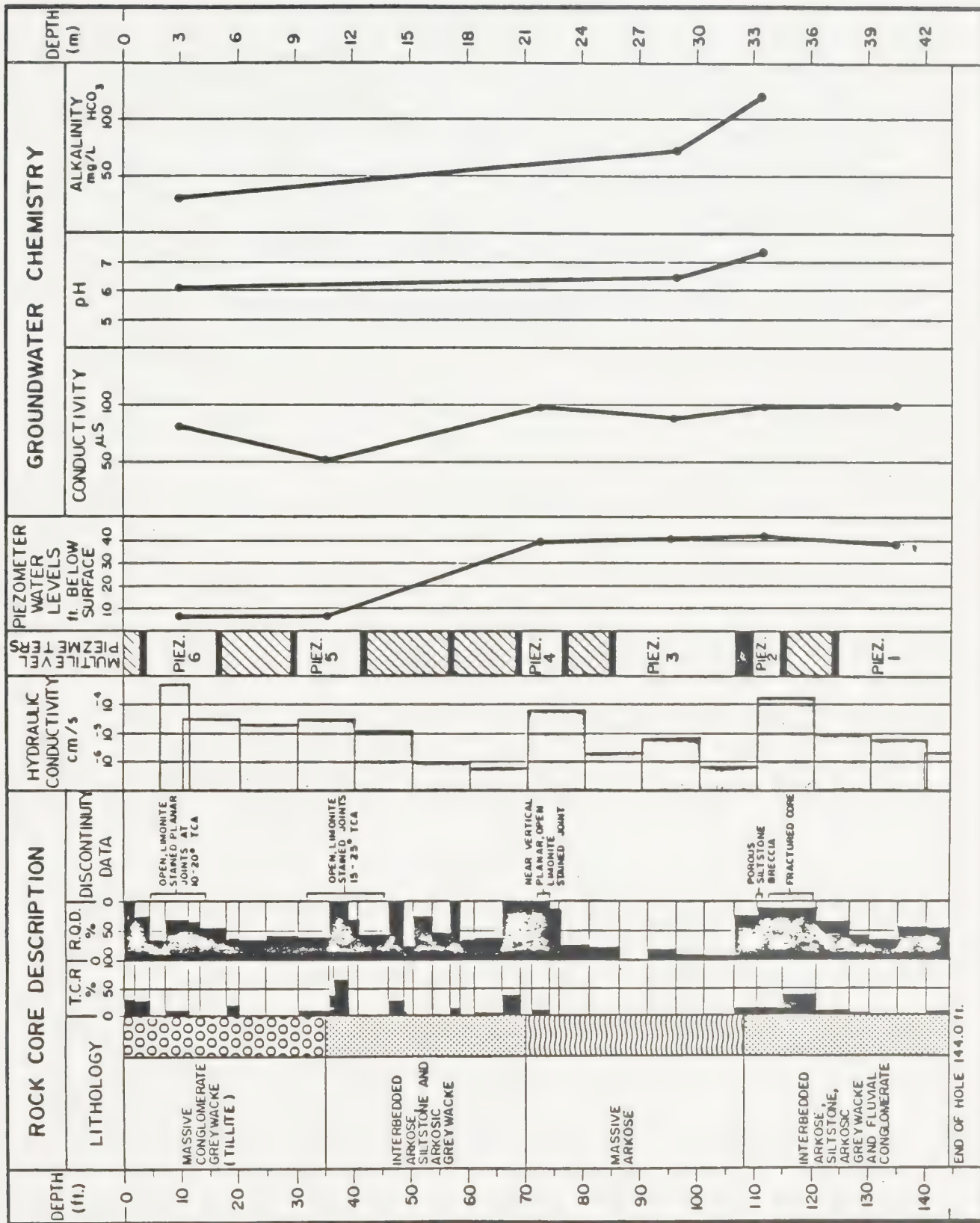


Figure 11. Depth profiles of hydrogeologic parameters at borehole 602 at the tailings impoundment site.

HYDROGEOLOGICAL INVESTIGATIONS OF
THE UPPER OTTAWA STREET LANDFILL SITE

PART 2

GROUNDWATER MONITORING AND PRELIMINARY
DESCRIPTION OF HYDROGEOLOGICAL CONDITIONS

ABSTRACT

The Upper Ottawa Street landfill exists in an area of complex topography that is generally underlain by a thin veneer of glacial overburden on bedrock. Part of the landfill has been constructed over the original channel of Redhill Creek. The bedrock consists of limestone, dolostone and shale.

During four phases of drilling from April to October, 1982, nineteen multilevel sampling devices were placed in vertical boreholes from which continuous core samples were obtained. The boreholes were placed around and beyond the landfill. Piezometers from a previous investigation exist within the landfill. The boreholes ranged in depth from 20 m to 45 m. The multilevel devices installed during the first three drilling phases were found to have minor leakage points, which apparently occurred along the interior of the packers or at the O-ring seal at the piezometer ports. In November, bentonite slurry was pumped into the centre columns of the PVC casings of these multilevel devices to seal the leakage points. Three multilevel devices installed during the last drilling phase contained modified types of packers and piezometer ports. These devices exhibited no leakage of water between the inside of the casing and the outer monitoring zones.

Examination of cores and the results of water-level response testing of piezometers in the multilevel devices indicates that the bedrock is slightly to highly fractured along horizontal bedding planes.

Borehole evidence and the geologic setting indicates that some vertical fractures are present, although the frequency of vertical fracturing is unknown because all of the drill holes were vertical. Water-level response tests of piezometers indicate that the bulk horizontal hydraulic conductivity of the rock is generally in the range of about 10^{-10} to 10^{-3} cm/sec. Many borehole segments in the shallow zones are in the upper half of this range, whereas segments in deeper rock zones generally exhibit lower hydraulic conductivities.

A local water table mound exists within or beneath the landfill. The regional water table slopes from the landfill towards Albion Falls and the Niagara Escarpment in this area. Because of the mound, it is expected that leachate moves radially outward from the landfill in the shallow part of the bedrock. Several of the multilevel monitoring devices near the landfill indicate strong hydraulic gradients from the shallow groundwater zone to deeper zones in the rock. This suggests that there could be a potential for movement of contaminated groundwater to a considerable depth in the rock. The high hydraulic conductivity in the shallow groundwater zone, however, suggests that the bulk of the contaminated groundwater moves laterally in this shallow zone. Without knowledge of the vertical hydraulic conductivity in the deeper zones, it is not presently possible to estimate the proportions of downward versus lateral flow.

Most of the multilevel devices have been sampled on two or more occasions for analysis of electrical conductance, chloride, and total dissolved organic carbon (TOC). These parameters have been used to monitor the progressive removal of drilling water from the sampling

zones as repeated episodes of pumping occur. High concentrations of both chloride and TOC are also indicative of gross contamination from the landfill. These parameters, however, are not well suited for identifying zones that are not grossly contaminated because the natural groundwater in the bedrock also contains considerable chloride and dissolved organic carbon.

Water samples were analysed for a number of inorganic parameters. Many exceed one or more MOE drinking water quality criteria, especially for ammonia, boron, chloride, chromium, iron, lead, manganese and sulphate. Even apparently uncontaminated groundwaters commonly exceed the MOE criteria for chloride and sulphate, reflecting the generally poor natural bedrock groundwater quality in this area. Preliminary organic analyses indicate these groundwaters apparently lack volatile, halogenated compounds such as common solvents. Other volatile and non-volatile organics are present; it is hoped that critical organic compounds can be identified which delineate landfill-leachate contamination at this site. Clearly, an integration of physical hydrogeology, inorganic, organic, and environmental isotope geochemistry will be required to define the nature, extent and probable impact of groundwater contamination.

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INTRODUCTION

The Upper Ottawa Street landfill is located near the Niagara Escarpment in Hamilton, Ontario, as shown on Figure 1. The landfill covers a large area within a complex topographic setting on fractured rock. From the hydrogeological studies of the site conducted previously by Gartner Lee Associates Limited (1979), it is apparent that a water-table mound occurs within or beneath the landfill. The presence of this mound, combined with the effect of the local topography and of fractures in the rock, provides for the possibility of subsurface contaminant migration in many directions at many rates.

Hydrogeological investigations of the landfill site were begun by the University of Waterloo in 1982, and are continuing in 1983. The primary objective of these investigations is to determine the extent of the zone of contaminated groundwater at the site and the chemical composition of this zone. The premise upon which these investigations are founded is that contaminant migration in the bedrock at the site occurs in many directions along complex pathways through fractures in the rock. Based on this premise, our strategy is to install a network of bedrock monitoring devices that will provide for groundwater sampling at many depths at numerous locations. For this purpose, the multilevel monitoring device described in Part 1 of this report was developed.

The locations for drilling and installation of the multilevel device were selected based on topographic setting, proximity to the

landfill and on the preliminary results obtained from previous studies. The expansion of the monitoring network proceeded in stages; several devices were installed initially, preliminary data were acquired and evaluated and then new sites were selected. Conceptually this approach is simple but in practice it is slow and tedious. Identification of groundwater containing landfill-derived contaminants has been difficult because uncontaminated groundwater in the bedrock commonly contains considerable concentrations of the chemical constituents that normally serve as indicators of landfill leachate. A second difficulty is the considerable disturbance of the hydrochemical conditions in the groundwater around the borehole that results from drilling cored holes in the rock. Relative to landfill studies in other hydrogeologic settings, much more effort and time is required to obtain samples representative of in situ conditions.

This part of the progress report describes the field investigations that were undertaken in 1982, and presents some preliminary interpretations of the data available by late February, 1983.

SITE DEVELOPMENT

An important aid in the interpretation of hydrogeological data from a waste-disposal site is the acquisition of information on the types and depositional locations of various wastes that entered the landfill. This is particularly true for the Upper Ottawa Street landfill because of its large size and because it received a variety of solid and liquid wastes during its lengthy use. There is a paucity of information on the nature of the wastes that entered the landfill because formal and continuous records were not kept during its life. Assemblage of information on this topic from diverse sources or records was not part of our hydrogeological study; this task is being pursued by the Upper Ottawa Street Landfill Site Study Committee.

To trace the development of the Upper Ottawa Street Landfill, all available stereoscopic aerial photographs were obtained from the Ontario Ministry of Natural Resources. Aerial photographs taken in 1954 and in 1972 show the region at a scale of approximately 1:15,000; the most recent photographs, taken in 1978, show the area at a scale of 1:10,000. Interpretations based on stereo viewing of these photographs are shown on Figure 2.

The 1954 photos show the early development of the landfill. Landfilling was restricted to a small area within the western part of the present landfill, shown as area (1) on Figure 2. Also shown on the 1954 photos is an old quarry, north of the present entrance gate and since filled with materials of an unknown nature.

Most of the expansion of the landfill took place between 1954 and 1972. Part of the expansion was accomplished by diverting Redhill Creek to the north and filling on top of the former flood plain. This diversion may be significant in that the former streambed could represent a relatively high-conductivity path for contaminant migration.

In area (2), filling appears to have been essentially completed by 1972, except for minor surcharge. The 1972 photos show filling in progress in area (3) adjacent to the railroad tracks along the east side of the landfill. Filling appears to have just begun in area (4), along the southern part of the landfill. In this area, landfilling was apparently preceded by excavation of surficial soils, probably for use as cover material elsewhere in the landfill.

The large area shown as disturbed on Figure 2 appears in the 1972 photographs. If any fill is present in this area, it is probably shallow and largely inert.

Very little lateral expansion of the landfill took place between 1972 and 1978. Some new filling occurred along the eastern part of the landfill (area (5) on Figure 2), but most of the landfilling was restricted to the topping off of Areas 3 and 4. The most significant features on the 1978 photos are some lagoons, presumably used for fixation and percolation of liquid wastes. These ponds are located near the former course of Redhill Creek. The two northwestern ponds contained liquids in 1978, and the southwestern pond appears to have recently contained liquid. The remaining ponds are indistinct in outline, and do not appear to have been recently used at the time of the photographs.

These aerial photographs provide a gross indication of the historical development of the landfill. It is likely that additional useful aerial and ground photographs exist in the files of engineering firms and government agencies; their acquisition will be pursued in 1983.

FIELD METHODS

Drilling and Coring

A tracked-vehicle mounted CME 75 drill was used for drilling and coring at 20 sites in the vicinity of the landfill. Each site was drilled by augering through the overlying clay till to bedrock using hollow stem (10.8 cm ID) augers. Casing was set approximately 0.6 m into the bedrock. Bedrock was then continuously cored using NVL core barrels of 1.5 m and 3.0 m lengths.

Water was used during coring of the bedrock, both to cool the drill bit and to remove the drill cuttings from the borehole. The drilling water, taken from local fire hydrants, is supplied from Lake Ontario. The water used during drilling was not recirculated in the hole as coring took place.

Core removed from each hole was logged in the field with respect to geologic and hydrogeologic characteristics. Locations of geologic units and fracture zones were recorded. Fractures were examined with respect to orientation, size, infilling, and staining. From examination of the core, zones with apparent higher hydraulic conductivities (i.e. larger or more frequent fractures) were selected for placement of sampling intervals in the multilevel monitoring devices. Selected cores were described in more detail in the laboratory.

Design and Installation of Multilevel Devices

This section describes the multilevel monitoring devices with respect to field assembly and installation. A detailed description of the multilevel sampling device is provided in Part 1 of this report.

From preliminary testing in November, 1981, it was felt that five water sample points would be the optimum number of points for each multilevel device. Thus, five fracture zones were generally chosen for each core hole and a multilevel device was designed and assembled based on the location of these zones. Multilevel devices installed during the early phases of drilling had water sampling zones varying in length from 0.6 m to 1.2 m, depending on the size of fracture zones encountered. However, for later phases of drilling, water sampling zones were generally 1.2 m in length because many fracture zones, especially in the shale, produced very small amounts of water, and it was decided that the longer sampling zone was necessary to obtain sufficient water.

Once water sampling zones were chosen, the multilevel device was assembled above ground in sections. Sample tubes and sample points were assembled for each sample zone as described in Part 1. Each sample point was located near the bottom of the sampling zone. This was done for two reasons; first, to facilitate attaching the sample point to the PVC casing and second, to enable removal of the entire column of water located in the sampling zone in order to obtain fresh formation water when sampling.

Packers were then coupled to the PVC casing for each sampling zone and sections of PVC casing were then coupled to the packers at appropriate intervals to correspond with the previously determined

sampling zone depths. If long intervals of PVC casing were present without any sampling zones, packers were placed in the intervals to prevent hydraulic connection along the outside of the PVC casing. The length of each coupled section was approximately 3 m to 4 m. This avoided cracking the joints under the weight of the sampling tubes when the section was lifted upright for placement in the hole. The bottom section was then placed in the core hole and held at ground surface while the next section was coupled to it. If sampling points were encountered, the sampling tubes were fed through the next section to be coupled and each new section coupled in the same manner. Sections were coupled by glueing with PVC glue. This process was repeated until the entire assembly was placed in the core hole.

The inside of the PVC casing was filled with water and after the packer was allowed to expand, water-level monitoring and sampling began.

Testing of the Multilevel Devices

Preliminary Testing

The first multilevel device was installed in November, 1981 and was tested briefly to ensure that the packers were performing properly. A packer joint was cracked at one of the water sampling points during installation and, as a result, testing of a well point with a known hydraulic leak could be done to determine if a leak could be easily detected in future installations.

This first device was tested in two ways; first, by response testing each well point and second, by chemical analyses of water

samples from each point for specific conductance, total organic carbon and chloride.

Water levels in each well point and the PVC casing were measured, and then each individual point was pumped while adjacent points and the inside of the PVC casing were monitored for changes in water levels. Water levels were different in each well point prior to pumping, with levels in the wells ranging from 1.45 m to 2.39 m below the top of the casing except for the deepest point which was flowing (the casing is approximately 1.0 m above ground surface).

During pumping of each well point no change in water levels was observed in adjacent points. On pumping the well point with the cracked packer joint, however, the water level in the PVC casing immediately declined. Water from this sample point was a distinct yellowish colour, characteristic of water in contact with the packer sealant.

Water chemistry was found to differ significantly in each well point. Specific conductances ranged from 4000 uS to 97,000 uS, total organic carbon ranged from 40 to 339 mg/L (except for the leaking well point), and chloride ranged from 1130 mg/L to 54,000 mg/L. The PVC casing water had a total organic carbon content of 3560 mg/L and chloride of 33 mg/L. The total organic carbon content of the well point which leaked was 3680 mg/L which comparable to the PVC casing water.

It was on the basis of this testing that it was decided that the packers had performed successfully, that major leaks could be identified, and further installation of multilevel devices would be continued at the Upper Ottawa Street Landfill in 1982.

Detailed Testing

Following installation of 19 multilevel devices at the Upper Ottawa Street Landfill during the spring, summer and fall of 1982, more detailed testing was done in the field to assess the performance of each device.

A solution of rhodamine dye was injected into the column of water in each of the PVC casings. At most installations, the dye solution was injected through a narrow diameter polyethylene tube placed inside the PVC casing as far as possible toward the bottom (generally 18 m to 23 m) in order to ensure the dye would mix through the entire column of water in the PVC casing. In some installations dye was injected at a shallow depth in the PVC casing water during routine water sampling.

The dye mixed readily, as indicated by bright red water that was initially displaced from the PVC casing of many installations. Installations in which dye was injected in the shallow zones were tested by pumping from a tube placed at a lower depth inside the PVC casing. Installations tested even one day after dye injection showed red water when pumped, indicating that the mixing of the dye throughout the PVC casing water was fast and effective.

At most installations, water levels were monitored in the piezometer tubes and in the PVC casing before, during and after injection of the dye. Fluctuations of water levels in response to the injection would indicate hydraulic connection between the PVC casing water and the piezometer, thus signifying leakage was occurring. After the dye solution was injected, piezometer tubes were flushed using the

sampling methods described in the next section. This flushing was done to determine if dye was present in any of the well points. Results of the testing are discussed in a later section.

Routine Monitoring

After the monitoring devices were installed, water levels were measured and water was pumped out of each piezometer several times. When the water column in each piezometer was removed, samples were collected for analysis of specific conductance, total organic carbon (TOC) and chloride. These parameters were selected because they can be determined quickly and inexpensively and were expected to provide a means of identifying gross contamination from the landfill. Specific conductance was measured in the field and TOC and chloride were analysed at the University of Waterloo.

Routine sampling was initially performed at each site because drilling involved addition of foreign water to the hydrogeological system and it was not known how long it would take before the effects of the drilling water would disappear at each piezometer. Because each piezometer is connected to fracture zones with different hydraulic conductivities, repetitive sampling of each well point was done to determine if and when the hydrochemistry had stabilized. A stable water chemistry during repeated sampling was taken as indicative of the actual groundwater chemistry for the fracture zone.

Groundwater samples were collected using two methods. Piezometers that yielded water continuously when pumped and that had shallow water levels (i.e. < 8 m) were sampled using a peristaltic pump

and small diameter polyethylene tubing that was placed down each sampling tube. Where sample zones could be continuously pumped, specific conductance was regularly measured to determine if stabilization of the water chemistry was occurring. It was expected that these well-waters would rapidly stabilize as the drill water was flushed out of the fracture zones. Water samples were collected for TOC and chloride at intervals during pumping to determine if these parameters were also stabilizing.

Piezometers that yielded water very slowly and those that had deep static water levels were sampled using a gas-driven sampler referred to as a triple-tube sampler, described by Robin et al. (1982). The sampler operates by displacing water under positive nitrogen pressure. This pressure displaces the entire water column from the sample tube. The water level in the piezometer tube is then allowed to recover, and water is again forced out and collected for analysis. In some piezometers the permeability of the rock is so low that it was necessary to wait several days or longer until the water had risen sufficiently for the water column to be displaced again.

Sampling at all wells involved the use of tubing inside each piezometer tube. To avoid cross-contamination of piezometers, the tubing was cleaned on the outside by drawing it up through a methanol-wet cloth and then rinsing with deionized water. The inside of the tubing was cleaned by pumping 300-500 ml. of deionized water through the tubing. The tubing was then placed in a new sample point tube by drawing it through a deionized water-wet cloth. Samples were collected in scintillation vials and kept cool until sent to the University of Waterloo for analyses for TOC and chloride.

Prior to sampling of each piezometer, water level measurements were taken using a narrow diameter electric probe made of coaxial cable. The end of the probe was cleaned with deionized water after each water level measurement to prevent cross-contamination of piezometers.

Water Sampling for Inorganic Analyses

In the fall of 1982, 27 groundwater samples from 21 piezometers were analysed for major dissolved constituents, nitrogen species, dissolved organic carbon and 11 trace constituents. Results to date are listed in Table 1. Also listed are pH and Eh (redox potential) measured on site, usually with a flow-through cell isolated from the atmosphere.

Samples were obtained with a peristaltic pump, which is a vacuum lift system, where static water levels were within about 8 m of the surface. Otherwise, the triple-tube sampler was used. Samples were filtered through 0.45 micron filters in the field and an aliquot acidified with nitric acid. Samples were kept cool until delivered to the MOE laboratory, usually within 1 or 2 days after collection.

Piezometers had been previously flushed, usually many times during the routine monitoring, and so were expected to be relatively free of drilling fluid contamination. In many cases, replicate-sampling yielded waters with very different chemistry (see Table 1). This has been partially addressed in the sampling protocol development discussions to follow, but clearly continued sampling will be required before the true groundwater chemistry can be established.

Water Sampling for Organic Analyses

Water samples were collected in the early fall for preliminary organic analyses at many of the same locations sampled for inorganic analyses. Water samples were collected using sampling techniques described in the section above. Samples were analysed qualitatively for purgeable organics at Waterloo. Additional samples were also collected at 8 locations, anticipated to be in highly contaminated groundwater zones, from Gartner Lee wells and University of Waterloo wells. These samples were obtained with an all-teflon-stainless steel-glass system to minimize contamination and loss of organics due to adsorption on plastic tubing, and were sent to Stanford University for analyses of a complete suite of organics. This was done in part to determine if a particular set of organic compounds could be found that would be indicators of the landfill leachate. A sampling and analysis program will then be established based on these results and results of inorganic analyses to attempt to define the contaminant plume.

RESULTS AND DISCUSSION

Hydrostratigraphy

The Upper Ottawa Street landfill is located on the plateau lands above the Niagara Escarpment. Geologically, this is an area of undulating glacial drift underlain by a bedded dolomite and shale bedrock. The geology is reflected by the local topography. The major topographic feature of the area, the Niagara Escarpment, is related to the presence of a cherty dolomite bed which is significantly more resistant to weathering than other dolomites and limestones in the sequence. A much smaller rock scarp is found southwest of the landfill, and trends northwest and southeast of the site. Although the bedrock surface along this scarp just west of the landfill is at the highest elevation found in the region, the scarp has been covered in the landfill area.

Knowledge of the general characteristics of the geologic units has been extended by drilling in the vicinity of the landfill. Twenty sites have been drilled to date by the University of Waterloo. Continuous rock core has been obtained from each site, and multilevel devices have been installed in all of the holes. Figure 3 shows the location of these twenty monitoring sites.

Prior to the commencement of the University of Waterloo study, monitoring wells were installed by Gartner Lee Associates Limited. These monitoring wells are shown on Figure 4.

Simplified borehole logs for the sites drilled by the University of Waterloo are shown on Figures 5 to 24. These borehole logs were used to construct geologic cross-sections through the landfill area, shown on Figures 25 and 26. Locations of the cross sections are shown in Figure 3.

The uppermost geologic unit consists of glacial drift. As encountered in the University of Waterloo borings, this drift comprises glacial till and glaciolacustrine clay. It is generally less than 3 m thick except in the moraine located north of the landfill, where the thickness is about 12 m.

Bedded dolomite and shale bedrock occurs below the glacial deposits. The uppermost portion of this formation in the vicinity of the landfill ranges from a shaly dolomite to a dolomitic shale. This unit probably corresponds to the Eramosa Member of the Lockport Formation as described by Bolton (1957). The unit is described as being a dense thin bedded dolomite that is petroliferous, with bituminous shale partings. The upper zone, which is more massive, could correspond to the less shaly dolomite encountered (i.e. UW5) during drilling. Although more massive, this unit has some intervals where significant fracturing is present. This was indicated during coring of UW5 by water cascading down the drill hole from shallow zones. Most fractures, however, were very tight, usually occurring along shale partings. The fractures were horizontal or nearly horizontal. As a result, much of this unit will generally have a very low hydraulic conductivity with one or two high conductivity fracture zones encountered at each site.

The presence of the bituminous shale has a significant effect on the groundwater chemistry. Deposition of most shaly rock occurred in salt water and some of the salt remains in the rock pores for a long period of time. Consequently, groundwater in the shale may contain high concentrations of dissolved solids such as chlorides. Because the shale in this unit is bituminous, it may also impart organic carbon to the water.

Below this unit is a cherty dolomite which is known as the Goat Island Member of the Guelph Lockport Formation and was encountered at every location drilled. This unit is a massive, irregularly bedded, fine-grained dolomite with many chert nodules and lenses at the base. This unit forms the cap rock for much of the Niagara Escarpment in the Hamilton area. The distinctive nature of this cherty unit makes it useful as a marker bed throughout the area.

The cherty dolomite encountered during coring at each location indicates that the unit is massive with few fractures. Fractures were generally horizontal or near-horizontal and occurred more frequently as fracture zones, rather than isolated fractures. Some vugs were also present and were filled with calcite and gypsum. One or two larger fracture zones were usually encountered at each site. These were the zones of highest hydraulic conductivity encountered in the study area, as evidenced by large water losses while drilling. The significance of these high hydraulic conductivity zones depends on the continuity of the fractures throughout the unit; the frequency of fractures would suggest a high degree of continuity through the rock mass.

Underlying the cherty dolomite is what has been described as a shaly dolomite in Figures 5 to 24. Although described as a single unit, this may in fact be two units as described by Bolton (1953). These are the Gasport Member of the Lockport Formation and the underlying Decew Formation. At the Upper Ottawa Street Landfill this unit has a highly variable lithology ranging from a semi-crystalline dolomite to finely-bedded shaly sections up to 1.0 metre thick. Alternating thinly bedded semi-crystalline and shaly zones were often encountered during drilling.

Very few fractures were encountered in this shaly dolomite unit. Most fractures were horizontal, occurring along the shale bedding planes and were very tight, indicating a relatively low hydraulic conductivity throughout this unit. The only significant fractures were found in the less shaly upper sections of this unit. These fracture zones did not appear to be as conductive as the zones encountered in the cherty dolomite. Because of the highly variable lithology within this unit, the groundwater chemistry is also expected to be highly variable.

Below the shaly dolomite is a shale unit known as the Rochester Shale. It consists of dark calcareous shale and siltstone with some thin limestone beds. The upper zone is generally a massive dolomitic or calcareous shale. The shale is slightly or moderately fractured, generally along the horizontal bedding planes. Many of the fractures are infilled by clay and occasionally calcite and gypsum. Some shale zones encountered were up to 2.0 metres thick without any noticeable fractures. The shale sections of this unit appear to have the lowest hydraulic conductivity of any unit encountered within the study area.

Limestone beds up to 0.3 m thick were encountered in the Rochester shale. These units contained some fractures which were generally horizontal, very tight, and did not appear to have any zones of high hydraulic conductivity.

The lowest unit encountered in some of the boreholes was a limestone underlying the Rochester shale. This is probably the Irondequoit formation, and is a massive, crystalline limestone. Although little core was available for examination, the fracture zones that were encountered may have a high hydraulic conductivity.

Leakage Detection and Sealing

Multilevel devices were field tested as previously described. Observations of dye indicate that some leakage of water from inside the PVC casing (centre column) into some sampling zones was occurring. Water level measurements taken in sampling tubes during dye injection showed water level changes in a few piezometers. Pumping of sampling zones after dye injection in the centre column showed dye present in a small percentage of the piezometers.

Changes in water levels in sampling zones during injection would suggest the water level in the sampling zone was responding to an increase in water level in the centre column, indicating a hydraulic connection between the centre column and the sampling zone. In the multilevel devices tested, the zones which showed changes in water levels were generally the low hydraulic conductivity zones (i.e. slow responding zones). This is probably because the fractures are small and it would take very little leakage of water to change the water

level. In contrast, high permeability zones would not necessarily demonstrate a change in water level because the effect of leakage could dissipate quickly.

In examining the results of pumping of piezometers after the dye was added to the water column in the casing, the following types of results were observed in different piezometers: 1) piezometer water remained without dye during the entire pumping period, 2) piezometer water initially showed dye but became clear after a short period of pumping, 3) the piezometer water was initially clear during pumping but showed dye after pumping for a period of time, or 4) the piezometer water exhibited dye during the entire pumping period.

For the piezometers that remained clear during the entire pumping period, it can be concluded that these piezometers do not have any leakage or significant connections with the inside of the casing. Many of these piezometers were those within the high hydraulic conductivity zones (i.e. the zones with highest water yield rates).

The second case indicates very slow leakage. As the water in the sampling tube is flushed, leakage from the centre column is not fast enough during pumping to provide a continuing visual presence of dye in the piezometer water. In the third case, the water level in the sample tube may be at a level that will not allow leakage from the centre column water until the sample tube is pumped, thereby lowering the water level enough to induce leakage in the sampling zone. This occurred in a few low hydraulic conductivity zones. In the fourth case, some percentage of centre column water was found in the sampling zones during the entire pumping period, indicating a continuous leak of centre column

water into the sampling zone. This was only found in a small number of piezometers.

Analyses of water in contact with packer material in the lab and centre column water from installations in the field indicate TOC to be in the range of 3300-3600 mg/L and chloride concentrations to be 30-35 mg/L. More complete inorganic chemical analyses of water in contact with Dowell sealant and other piezometer materials are presented later. Although numerous cations and anions can be leached from piezometer materials and especially from the sealant material, there is no indication that groundwaters analysed to date suffer serious contamination. The leaching water contained significant 1,1,1-Trichloroethane but only traces of other volatile, halogenated organics. Again, there is no evidence of significant contamination by this compound in samples analysed to date. Analyses of other leachable organics are continuing.

Due to the shaly nature of much of the rock in the study area, the groundwater chemistry is expected to have low to moderate TOC and high chloride concentrations. As a result significant leaks should be easily detectable by routine analyses. Many of the sampling zones which showed some indications of dye during leakage tests routinely had chemical analyses showing TOC as low as 25 mg/L and chloride concentrations as high as 40000 mg/L. This indicates that these piezometers were only influenced to a minor degree by leakage from the centre columns.

The multilevel device was further tested in the laboratory to determine whether the points of leakage could be detected. Segments of

PVC casing separated by packers were placed in 7.5 cm (3 in) ID plexiglass tubes and the centre column was filled with water to determine the quantity and location of leaks. It was found that when leakage was occurring in the sampling zone it was generally at the point where the elbow on the piezometer port protruded through the casing. Some packer leakage occurred during the initial expansion of the Dowell seal sleeve. The seals became water tight after a few days.

Both of these problems have since been rectified by changes in the design of the piezometers that were installed later in the drilling program. Small glued PVC ports are now being used instead of previous O-ring type sample ports. To prevent the initial leakage of water along the packers, double clamps have been used and silicone sealant placed between the clamps. These modifications are discussed in more detail in Part 1.

The results of leakage testing described above pertain to the first 17 multilevel devices installed at the Upper Ottawa Street landfill site. The last three devices that were installed in the fall of 1982 contained double packers at each packer interval and each packer had silicone-sealed, double stop clamps around the rubber sleeves on the packers. These three multilevel devices also contained glued elbow joins rather than the O-ring type joins that were used in the previous devices. Monitoring of the water levels in the interiors of the PVC casings in these devices indicates that there is no leakage from these casings.

The final stage in the preparation of the multilevel devices for stable, long-term monitoring involved filling the centre column of

each of the devices with a bentonite slurry. The purpose of the bentonite slurry is to seal any leakage points that may exist at the couplings, at the piezometer elbows or along the inside of the packers. This stage, in which the first 16 multilevel devices were filled with bentonite slurry, was conducted in November, after all of the multilevel devices that were planned for 1982 had been installed.

The bentonite slurry was created by mixing powdered bentonite (Quick Gel) with water to an optimum viscosity suitable for pumping through flexible 1.5 cm OD polyethylene tubing. The tubing was inserted to the bottom of the PVC casing. The bentonite slurry was pumped down the tubing using a small centrifugal pump. As the bentonite slurry enters the casing, it displaces the water column upward. Pumping of the slurry continues until all of the water is displaced, at which time the slurry flows from the top of the casing. As the slurry ages in the casing, it transforms from a watery slurry to a soft clayey mass, which prevents movement of water into or out of the casing or upward or downward within the casing.

Physical Hydrogeology

Water Levels and Groundwater Flow

During the present study, detailed monitoring of water levels in the University of Waterloo installations was performed to gain some understanding of groundwater flow directions in the vicinity of the landfill. Because the principal emphasis during this initial phase of the study was on sampling the piezometers as frequently as possible, water levels measured in some piezometers were never able to attain



equilibrium. At most piezometers, however, there were instances when the piezometers were allowed to remain undisturbed for a sufficiently long interval to permit a number of equilibrium measurements. Figures 30 through 49 show the water level hydrographs for installations UW1 through UW20, respectively.

Figure 27 shows a generalized water table contour map. This map is based primarily on August, 1982 water levels measured in the University of Waterloo multilevel devices. In areas where University of Waterloo multilevel sampling devices do not exist, but where data are available from Gartner Lee wells, water level data from early September of previous years for the Gartner Lee wells were used. Such areas include the landfill itself and small areas north and south of the landfill. The water table contour map is therefore a generalized representation of late summer water level conditions. Although water levels generally rise and fall seasonally, the hydrographs on figures 30 through 49 indicate that during the present monitoring period there have not been any significant water level fluctuations that appear related to seasonal effects. As a result, it is unlikely that the general form of the water table changes appreciably.

Few water level data are available within the landfill itself. Two boreholes were placed within the landfill and monitored by Gartner Lee Associates. Water levels within these boreholes fluctuated as much as 2.0 m with no apparent trend. Data from these wells suggest that the water table within the landfill has mounded significantly. Mounding beneath landfills is a common feature in Ontario. This mounding is caused by greater infiltration and less evapotranspiration in the

landfill area relative to areas away from the landfill. A clay cover has been recently placed on the landfill to reduce infiltration. The effectiveness of this cover as a means of limiting infiltration is not known; it will probably be a considerable time before the groundwater mound dissipates noticeably. Recent water level monitoring conducted during the present study indicates that the mound has not begun to dissipate yet, two years after the clay cover was placed at the landfill.

With the exception of the groundwater mound, the water table map demonstrates that the lateral component of shallow groundwater flow is generally from west to east, with flow primarily directed towards Albion Falls Ravine to the east. An exception to this general easterly flow occurs north and northeast of the landfill, along the high clay till ridge (UW16 and Gartner Lee GL5 and GL6), where there appears to be a water table divide in which shallow groundwater flows towards Redhill Creek on the west and southwest side of the divide. Redhill Creek acts as a groundwater discharge zone within the shallow groundwater flow system.

Hydrogeologic sections shown on Figures 28 and 29 show vertical profiles of water level data along cross sections A-A' and B-B', respectively. Figures 5 through 24 show representative vertical profiles of hydraulic heads for UW1 through UW20. The vertical profiles and hydrographs (Figures 30 to 49) indicate a number of trends with respect to groundwater flow in the vicinity of the landfill.

In the shallow groundwater zone adjacent to and downgradient of the landfill, the vertical gradients are generally very small. This

shallow zone is within the cherty dolomite and overlying shaly dolomite, which corresponds to the zone of highest detected hydraulic conductivity, as discussed in the next section. Below this zone, in the low hydraulic conductivity shale, the hydraulic gradient generally has a strong downward component. The small vertical hydraulic gradients in the shallow groundwater zone and much larger gradients in the underlying shale are demonstrated by data for UW6 through UW9, shown as vertical profiles on Figures 10-13 and as hydrographs on Figures 35 through 38.

Hydraulic Conductivity

Values of hydraulic conductivity were determined for 50 piezometers at 13 of the multilevel sites and at 9 Gartner Lee monitoring sites. For each piezometer in the multilevel devices the calculated values represent the bulk hydraulic conductivity for the borehole interval from the packer below the piezometer port to the packer above the port. The hydraulic conductivity values are not assigned to individual fractures. For the Gartner Lee piezometers, the hydraulic conductivity values relate to the borehole interval from the bottom to the top of the sand pack.

Hydraulic conductivities were determined by monitoring the rate of water-level rise in piezometers after water levels were drawn down by pumping or bailing. Hydraulic conductivities obtained in this manner represent the capability of the fractured rock in the immediate vicinity of the borehole and adjacent to the packed-off intervals to yield water.

The method of Hvorslev (1951) was used to analyse the water-level recovery data. To determine hydraulic conductivities using the Hvorslev method, the rate of water level recovery and the shape of the piezometer intake zone are utilized in the following equation:

$$K = r^2 \ln (L/R)/(2LT_0)$$

where r is the radius of the piezometer, L is the length of the intake zone, R is the radius of the intake zone, and T_0 is the basic time lag. The basic time lag is defined as the time that would be taken for complete water level recovery to occur in a piezometer if the initial rate of water inflow were maintained. T_0 is the value of time required to achieve 63% of the complete water level recovery.

The data are plotted on a semi-logarithmic scale of time versus relative water level recovery, as shown in Figure 50. In theory, the data should represent a straight line passing through the origin, however, a number of factors can cause a deviation from this ideal response. Figures 50 (a), (c), and (d) show examples of water-level recovery graphs that conform quite closely to a straight line, whereas Figure 50 (b) exhibits a non-linear response. This non-linearity may be the result of a decrease in the hydraulic conductivity of a fracture that connects to the test interval of the borehole or it may be due to the termination of one or more fractures near the borehole.

The hydraulic conductivity depends on the number of open fractures in each packed-off interval, the apertures of the fracture and the roughness of the fracture surfaces. Because the boreholes are vertical and because the main trend in fractures is along the bedding planes that are horizontal or nearly horizontal, the hydraulic

conductivity values from the water-level response tests are primarily indicative of the hydraulic conductivity in the horizontal plane. The hydraulic conductivity in the vertical plane may be significantly smaller than that in the horizontal plane.

Table 2 shows the results of the hydraulic conductivity tests conducted to date. In a number of tests the recovery of the water level was too fast to permit water-level measurements during the recovery period. As a result, a short recovery time was used to calculate the hydraulic conductivity, and the hydraulic conductivity of these intervals may in fact be greater than the values presented in Table 2. The values of hydraulic conductivity range from a minimum of 2.7×10^{-10} cm/sec to a maximum of 1.9×10^{-3} cm/sec. The minimum value represents an interval of rock that is almost impermeable, presumably because this interval contains very few open fractures. The maximum value represents an interval of rock that is quite conductive because of open fractures within the interval. Figure 51 shows the variation in hydraulic conductivities with elevations and for different rock types.

General trends can be observed from the hydraulic conductivity results obtained to date. The highest hydraulic conductivity occurs in the cherty dolomite and the upper zones of the overlying shaly dolomite, both of which are well connected hydraulically to the landfill (Figure 25). Most of the hydraulic conductivity values determined for these units are greater than 1.0×10^{-4} cm/s.

Shaly dolomites and shales occur below the cherty dolomite (Figure 25). These geologic units have significantly lower hydraulic conductivity than the overlying units. The results for the shaly

dolomite and shales also show a decrease in hydraulic conductivity with depth. The hydraulic conductivity in these units generally ranges from 10^{-7} - 10^{-9} cm/s.

These tests only give an indication of horizontal hydraulic conductivity. It is expected that the vertical hydraulic conductivity is significantly lower. The hydraulic conductivity data support our conclusion that the bulk of groundwater flow is lateral within the shallow cherty dolomite and overlying shaly dolomite, as previously discussed. In 1983, hydraulic conductivity values will be obtained for the piezometers that have not yet been tested.

GROUNDWATER FLUX AND VELOCITY

The flux of leachate from the landfill can be estimated by two methods. The first method, which was used by Gartner Lee Associates (1979), is based on water budget calculations pertaining to infiltration on the landfill surface. It is assumed that the amount of leachate that leaves the landfill by seepage is equal to the water entering the landfill by infiltration. The second method consists of estimation of lateral groundwater flow for the shallow permeable zone in the cherty dolomite and the shaly dolomite zone at the periphery of the landfill. It is assumed that the leachate that enters the lateral flow zone in the upper part of the bedrock near the bottom of the landfill represents essentially all of the leachate that emanates from the landfill.

The water budget method used by Gartner Lee Associates is based on a comparison of average precipitation and evapotranspiration. From a 20 year record of climatological conditions at Mount Hope Airport, the average annual precipitation was estimated to be 80 cm (31.3 inches) and the average annual evapotranspiration was estimated to be 62 cm. For a vegetated soil, the difference between the precipitation and the evapotranspiration, which represents the average annual potential water surplus, would be 18 cm (7.1 inches). Because the cover on the landfill is not vegetated, Gartner Lee Associates believed the water surplus would be larger than this calculated value. They selected a value of 28 cm (11 inches) and assumed that this amount

of water, which is about 35% of the annual precipitation, infiltrates into the landfill over the entire landfill surface. This is equivalent to 12.7 L/min per hectare (0.5 gpm per acre). For the entire landfill, this amounts to 1.2 L/s (30 gpm).

In the second method the lateral flow of shallow groundwater away from the land is estimated using the Darcy equation,

$$q = KiA$$

where q is the groundwater flux per unit cross sectional area, K is the hydraulic conductivity and i is the hydraulic gradient. The flux calculation is made for a cross sectional area (vertical plane), which is 10 m deep and 700 m long. This distance represents the length of the northern and eastern boundary of the landfill along Redhill Creek. The lateral hydraulic gradient along the boundary is approximately 0.08. The values of hydraulic conductivity of the shallow bedrock zone (i.e. in the upper 10 m) are generally in the range of 1×10^{-4} to 2×10^{-3} cm/s. With these values the Darcy equation yields a total lateral flux in the range of 0.6 to 11 L/s. The total flux estimated by Gartner Lee Associates using the water budget method (2.3 L/s) is within this range.

The flux estimates obtained from both methods have considerable uncertainty. In the water budget method the main uncertainty pertains to the percentage of surplus water that leaves the landfill area by surface runoff. No measurements of surface runoff have been made at the Upper Ottawa Street landfill site or at any other landfill site with a similar surface. Surface runoff estimates for the Upper Ottawa Street landfill are even more problematic than would normally be the case because the landfill surface has gradually changed

during the past few years as cover material has been applied and as contouring and small-scale erosion channels have developed.

There are several causes of uncertainty in the results of the groundwater flux estimates obtained from the Darcy equation. Groundwater from the landfill may flow deeper than the 10 m depth of the cross section. There are probably local zones on the cross section that have a higher or lower hydraulic conductivity than the range used in the calculation.

The water budget estimate and the Darcy-equation estimate are not directly comparable because some of the leachate emanating from the landfill is collected in the drain and sumps and because some of the leachate seeps into Redhill Creek at the edge of the landfill. Some of the leachate flows downward into deeper zones below the cross section. Despite the uncertainties in the results from both methods, it is reasonable to expect that the flux of leachate-contaminated groundwater away from the landfill is in the range of a few tenths of a litre per second to several litres per second.

In studies of contaminated groundwater the velocity of groundwater flow is normally estimated from the relation,

$$v = Ki/n$$

where v is the average linear groundwater velocity and n is the porosity. For fractured rock n represents the percent of a unit volume of rock occupied by open fractures through which groundwater flow occurs. Larger values of n are expected to be associated with rock that has larger values of hydraulic conductivity.

In fractured rock it is very difficult to obtain accurate representative values of n . Measurements of n have not yet been made at the Upper Ottawa Street site. For the shallow bedrock zone of highest hydraulic conductivity (1×10^{-4} to 2×10^{-3} cm/s), n may be about 1%. A representative value for the lateral hydraulic gradient in the shallow bedrock in areas downgradient of the landfill beyond the influence of the water-table mound in the landfill is approximately 0.025. These parameter values in the equation presented above yield a groundwater velocity range of 40 to 1600 m/yr.

These values indicate that there is considerable potential for landfill-derived contaminants to migrate large distances in the shallow bedrock zone. The actual migration distances, however, cannot be determined from calculations such as those presented above because the uncertainties in the values of K and n are large. To determine the extent of contaminant migration, it will be necessary to identify and map chemical constituents in the groundwater that are diagnostic of landfill leachate. Velocity estimates such as those presented above based on estimates of hydrogeologic parameters serve as a means of establishing some possibilities regarding the size of the area in which landfill-derived contaminants may occur.

WATER CHEMISTRY

Electrical Conductance, Chloride and Total Dissolved Organic Carbon (TOC)

Results of routine monitoring for chloride and total dissolved organic carbon (TOC) are shown in Figures 5 through 20 for installations UW1 through UW16, respectively. Results have not yet been obtained for UW17 through UW20. Only those results which indicate chemical stabilization are shown; these results form the basis for our analysis of possible zones of contamination.

The results of routine sampling show that concentrations are related to location of the installation with respect to the landfill, and also vary with the different geologic units. Concentrations of electrical conductance, chloride, and TOC are generally lower upgradient of the landfill than downgradient from the landfill.

In the cherty dolomite upgradient of the landfill, chloride values were generally less than 1000 mg/l (Figures 9 and 14). TOC values were in the range of 50 to 100 milligrams per litre and electrical conductance values were generally less than 1000 uS. The quality of groundwater in the shale and shaly dolomite upgradient of the landfill was somewhat poorer than that in the cherty dolomite. Concentrations of chloride were as high as 30,000 mg/L, but generally they range from 1000-8000 mg/L (Figures 9, 14 and 15). TOC values ranged from 60 to 200 mg/L, with a few values as low as 12-20 mg/L, and

electrical conductance values ranged from 900 to 3500 uS, although a few piezometers showed values in excess of 45,000 uS.

The piezometers that showed the highest concentrations of routine monitoring parameters are located adjacent to and east of the landfill along Redhill Creek and Redhill Creek Valley. These locations correspond to the inferred direction of groundwater flow from the landfill. UW1 (Figure 5), UW6 (Figure 10), UW7 (Figure 11), UW8 (Figure 12), and UW13 (Figure 17) have the highest concentrations of chloride. UW2 (Figure 6), UW3 (Figure 7), UW4 (Figure 8), and UW12 (Figure 16) also have high levels of chloride relative to those found in other multilevel devices. All these piezometers are located downgradient of the landfill site. The highest levels of TOC for those multilevel piezometers sampled were found in UW6 (Figure 10), UW12 (Figure 16), and UW13 (Figure 17); elevated levels were also found in UW2 (Figure 6), UW3 (Figure 7), and UW14 (Figure 18).

Downgradient from the landfill, chloride concentrations of up to 50,000 mg/L was found in the cherty dolomite (Figure 5); in the shale and shaly dolomite downgradient from the landfill, chloride concentrations were as high as 80,000 mg/L but generally ranged from 20,000-50,000 mg/L.

Results of routine monitoring also indicate that there may be movement of contaminated groundwater toward greater depths in areas downgradient from the landfill. UW6 (Figure 10), UW7 (Figure 11), UW12 (Figure 16), and UW13 (Figure 17) show increasing levels of chloride or TOC in the shallow groundwater zone. No change occurs at depth, while farther downgradient at UW2 (Figure 6) and UW3 (Figure 7) there is a

continuous increase in chloride or TOC concentrations with depth. This increase appears to be independent of rock type. Upgradient of the landfill there is no apparent trend toward increasing concentrations with depth as shown at UW5 (Figure 9) and UW10 (Figure 14).

One of the conclusions of the present study is that chloride, electrical conductance, and TOC may be of much more limited usefulness as indicators of landfill contamination in fractured shale and dolomite than in sandy geologic materials. Although these parameters have been routinely used as evidence of landfill-derived contamination in unconsolidated sand and gravel aquifers, because of the highly variable nature of the rock types underlying the Upper Ottawa Street landfill only general trends can be identified from these data. Contaminant migration from the landfill can best be defined by evaluating the results of these indicator parameters in conjunction with other chemical data.

Another conclusion of the routine monitoring study pertains to the time required for removal of drill water. As drill water is gradually removed by repetitive pumping, Cl^- , EC and TOC should exhibit more consistent values. Sample zones which stabilized the most rapidly were those which yielded at the greatest rates when pumped. Many of these zones were pumped continuously. Although, because of their high hydraulic conductivity, significant amounts of drill water may have entered these zones during drilling, this water appears to be quickly flushed out of the sampling zone by natural groundwater flow. Even after removal of all the drill water, the chemistry of the water in these zones may still vary significantly as a function of the length of

time the zone is pumped and the total volume of water removed when sampling. As pumping progresses, these variations may occur because the water being sampled is drawn into the piezometer from increasingly more distant fracture zones where water of different chemical composition may exist.

Sample zones which were slowest to stabilize are those low hydraulic conductivity zones which yield very little water when pumped. Results to date suggest that many piezometers have not yet stabilized. The rate of natural flushing of these zones is slow. Although only small volumes of drill water would have entered these low hydraulic-conductivity zones during drilling, only very small volumes can be removed each time piezometers in these zones are sampled.

Figure 52 shows examples of piezometers that exhibit rapid and slow chemical stabilization. The top two sampling zones (UW2-1 and UW2-2) show rapid stabilization; these are located in the high hydraulic conductivity cherty dolomite that was pumped continuously. The second to the bottom piezometer (UW2-4) had not stabilized after being sampled several times. This piezometer is located in a low hydraulic conductivity shale zone that could not be pumped continuously.

Inorganic Chemistry of Groundwater

Determinations were made of a major suite of inorganic constituents including calcium (Ca), magnesium (Mg), sodium (Na), potassium (K), bicarbonate (HCO_3), fluoride (F), chloride (Cl), sulphate (SO_4), iron (Fe), manganese (Mn), various forms of nitrogen (N) including nitrate (NO_3), nitrite (NO_2), ammonium (NH_4), and organic N,

and the trace metals cadmium (Cd), chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), selenium (Se) and zinc (Zn). In addition, pH, oxidation-reduction potential (Eh or pe) and temperature were measured in the field (usually with a flow-through cell closed to the atmosphere). Results are presented in Table 1. Sampling points selected were those that had no indication of leakage (dye-test) and were capable of producing considerable water. The sampling points were therefore situated in the relatively permeable zones. The selected points represent a good range of depths and geographical locations with respect to the landfill but no groundwater samples from within or directly below the landfill were collected. Multilevel monitoring devices have not yet been installed through the landfill.

Significant quantities of inorganic constituents can be leached from piezometer material if the Dowell sealant is exposed to the leaching water. Actual levels of leached inorganics will depend on packer material/water ratios, but for the particular leaching conditions used, the deionized water contained the following concentrations (mg/L) after about three months of exposure to the sealant and other materials:

Na - 670, K - 3.1, Ca - 6.9, Mg - 0.7, Fe - 1.8,

Mn - 0.05, Cl - 180, SO_4 - 902,

NO_3 - 14.5, HCO_3 - 160, pH - 6.8.

Most of these species are found in groundwaters at the Upper Ottawa landfill site, but NO_3 and Fe are usually found in much lower concentrations. In short, none of the chemical analyses presented in Table 1 indicate significant contamination by the Dowell sealant or other piezometer materials. This supports the conclusion based on trace, volatile organic analyses discussed later.

Groundwater sampling is not yet adequate to discuss the extent of leachate contamination. In particular, the lack of reliable samples of uncontaminated groundwaters from depths exceeding 11 m and the probable variability of natural groundwater quality in the bedrock precludes detailed discussion of the inorganic contamination. Repeated analyses of groundwaters from piezometers UW1-2, 1-4, 2-1, 3-4, 9-1, and 9-3 does not generally reveal consistent water chemistry. Even piezometer 1-2, which shows reasonable stabilization for chloride shows considerable variation in other species such as Ca^{2+} , NO_3^- and Fe. This could reflect residual contamination from piezometer installation and/or the natural variations in water masses being pumped from the low storage, fractured rock mass. Perhaps variations at piezometer UW1-2 are an example of the latter cause. Repeated sampling and extension of the network will be undertaken in 1983. Groundwaters sampled to date are slightly acidic to neutral (pH 5.64 to 7.02), moderately to strongly reducing (Eh from + 75 mV to - 167 mV) and extremely variable in inorganic composition. Based on the preliminary hydrogeological estimates of groundwater flow directions, samples from UW15-1, 15-2 and perhaps UW10-1 should not be affected by landfill leachate and are taken to representative of background water quality. Unfortunately, these sampling points are at depths of less than 11 m and are unlikely to be representative of deeper bedrock waters. Other groundwaters could be leachate-contaminated and many of these contain significantly higher levels of almost all inorganic species and TOC.

Ministry of the Environment water quality criteria for public water supplies are clearly exceeded for a number of parameters in a

number of piezometers. These include ammonia (NH_3), boron (B), chloride (Cl), chromium (Cr), iron (Fe), lead (Pb), manganese (Mn), and sulphate (SO_4). The permissible criteria (in mg/l) are: NH_3 - 0.5 as N, B - 1.0, Cl - 250, Cr - 0.05, Fe - 0.3, Pb - 0.05, Mn - 0.05, and SO_4 - 250. Samples from piezometers 15-1 and 15-2 exceed the guideline criteria only slightly for NH_3 and significantly for Cl and SO_4 . It seems likely that this does not represent groundwater contamination but rather poor natural groundwater quality in these rocks. The other groundwaters which exceed the guideline criteria are adjacent to the landfill and to the east of it and may reflect the migration of contaminants from the landfill site. Apparent groundwater contamination is not consistent for all inorganic species. For example, groundwater from UW13-5 greatly exceeds the criteria for Cl, NH_3 , Mn, significantly exceeds the criteria for SO_4 and B, but meets the criteria for Cr, Fe and Pb. On the other hand, the sample from UW9-3 collected on 03/11/82 greatly exceeds the criteria for Cl, NH_3 , Fe, Mn and Cr, significantly exceeds the criteria for B but meets the criteria for SO_4 and Pb. Clearly, the apparent inorganic contamination is complex and variable.

Although many piezometer-volumes of water were removed before the water samples were collected, there is considerable uncertainty about the most appropriate piezometer flushing and sampling protocol for obtaining groundwater samples. In deposits such as sand or gravel, removal of three to eight well volumes is thought to be an appropriate compromise between flushing of water standing in the piezometer and drawing in groundwater from more distant locations in the aquifer. In low-matrix-permeability, highly-fractured rocks, this flushing may

result in drawing in of distant groundwater and the sample would not be representative of the piezometer point. A project was undertaken (J. McKee, B.Sc. thesis) to evaluate the changes in groundwater quality during the removal of up to twelve standing piezometer volumes from UW9-1 and UW10-1. A peristaltic pump drew water to the surface where electrical conductivity, pH and Eh were measured. Samples were returned to Waterloo and to the MOE laboratory in Toronto for chemical analyses. Results are shown in Figures 53 where selected parameters are plotted over the number of well volumes removed prior to the particular sampling. Well volumes for UW9-1 and UW10-1 are 3.14 l and 3.24 l, respectively. Groundwater from UW9-1 is probably contaminated by landfill leachate, whereas groundwater from UW10-1 appears to be uncontaminated or only slightly contaminated.

For UW9-1, chloride is the dominant anion and so a good correlation between specific conductance (electrical conductivity) and chloride was anticipated. Figure 53 indicates a poor correlation up to about 1 well volume and then a fair correlation up to about 8 well volumes. The TOC concentration is extremely variable, while pH seems to stabilize to about 6.80 after about 5.5 volumes. Therefore, for piezometer UW9-1, stability for all but pH does not seem to have been reached within 12 well volumes of flushing.

Some additional parameters are included in Figure 53 for UW10-1, but only about 6 well volumes were pumped from this well. Changes in the bulk chemistry of the sampled groundwater are evident. Once again, TOC varies in a fashion not predictable from either chloride or specific conductance values. UW10-1 showed relatively stable pH

values at 6.80 after 3 well volumes as did UW9-1. On the other hand, Eh was rather unstable with values ranging considerably about a possible mean of + 170 m (mV) after about 1 well volume was removed. The concentration of iron, an Eh and pH sensitive parameter, was also unstable, perhaps about a mean of about 0.4 mg/l. After about 1 well volume was pumped, the iron trend was somewhat sympathetic to the Eh trend (Figure 53). This similarity was not expected since iron is usually found at higher concentrations in groundwaters with lower Eh values (i.e. more reduced waters). The apparent co-variance of Fe and Eh indicate that at least two waters are mixing; one high in both Fe and Eh and the other lower in both. Magnesium concentrations are more uniform at about 62 mg/l.

Although these studies are preliminary, the following observations are significant:

1. Few parameters co-vary. If stability in a chemical parameter is taken as indicative of a representative groundwater being obtained, then stability of the specific parameter(s) of interest must be confirmed. Field measurements of specific conductance, for example, do not define the trend in chloride concentration. Only direct measurements are dependable.
2. Large variations are apparent during initial (< 1 well volume) water withdrawal and are probably due to flushing of stagnant water from the sampling tube. Significant but smaller variations occur for most parameters up to 12 well volumes. These could be due to mixing of chemically-different groundwaters due to overpumping. Removal of large volumes of groundwater will therefore not provide stabilization nor a representative sample.

3. Flushing of perhaps two or three well volumes seems to provide a reasonable compromise.
4. Additional, longer-term experimentation is required but is unlikely to provide a protocol that is satisfactory for all piezometers at this site.

Methane gas, produced during anaerobic decomposition of organic matter, is emanating from the landfill. It is expected that methane will migrate above the water table as a result of gas flow and diffusion and that it will also migrate as a gas dissolved in leachate-contaminated groundwater. Methane may therefore be a useful indicator or tracer of leachate-contaminated groundwater. Consequently, the concentration of methane dissolved in groundwaters at twenty-four piezometer points was measured; at some piezometers it was measured on two or three occasions. The samples were collected in 50 cm³ glass syringes and they were analyzed by a gas chromatographic technique. Results are shown in Table 3.

Temporal variability of dissolved methane was observed, even at considerable distance from the landfill. UW3-4 was sampled on October 8, November 27, and December 3 and the methane (CH₄) content was 8, 22 and 25 cm³ per litre, respectively. Adjacent to the landfill (UW7-1), up to 30 cm³/L CH₄ is found, indicating some highly contaminated groundwaters are saturated with methane. The methane distribution is irregular, both laterally and vertically. This probably reflects at least in part the dual sources of methane in this area. Methane is a common gas in these bedrock formations, with some natural gas deposits present elsewhere. This methane is termed petrogenic or

thermocatalytic. Preliminary studies suggest that landfill-derived methane may be distinguished from petrogenic methane on the basis of its stable carbon isotope ratio ($^{13}\text{C}/^{12}\text{C}$). Landfill methane is less-enriched in ^{13}C than petrogenic methane. Additional sampling with isotopic analyses of methane will be undertaken in 1983 to evaluate the extent of landfill-derived methane in groundwaters.

Organic Groundwater Chemistry

The potential presence of industrial organic chemicals and the high organic carbon concentration in leachate-contaminated groundwaters has raised considerable concern about the potential migration of organic contaminants in groundwater from this landfill. Because of the analytical difficulties and the expense of trace organic analyses of groundwaters, only preliminary studies have been conducted to date. When the groundwater flow system and extent of inorganic contamination is better defined, more extensive studies of the organic constituents in the groundwater will be undertaken. This will likely occur during the summer of 1983. Volatile, halogenated solvents are among the most common environmental contaminants. They include chloroform (CCl_3), trichloroethanes (1,1,1-TCE and 1,1,2-TCE), carbon tetrachloride (CCl_4), trichloroethylene (TCEY), bromodichloromethane (BrCl_2Me) and bromoform (CBr_3), and many other halogenated methanes, ethanes and ethylenes. Table 4 presents the analyses of common volatiles performed by Stanford University, Department of Civil Engineering. The pentane extraction procedure outlined in Reinhard and Barker (in preparation) was employed. Specific organic concentrations did not exceed 0.2 ppb in any of the

groundwaters or in water seeping from the landfill face at Redhill Creek near UW7. For reference, drinking water standards of 10 to 100 ppb levels are likely to be introduced in many jurisdictions. To date, there is no indication of major groundwater contamination by volatile, halogenated organics at this site. It must be pointed out, however, that most of these organics have densities greater than normal groundwaters. This could cause such organics, if introduced in concentrated forms, to migrate downwards as a separate organic phase, (Reinhard and Barker, in preparation). In this case, the organics could exist as a discrete organic phase at depth below the landfill site. Continuous dissolution by moving groundwaters could cause deep groundwaters to become contaminated by these dense organics. Additional sampling for volatile organics is planned, with samples of deeper groundwater included.

As previously discussed, piezometer materials, including the Dowell sealant, were placed in distilled water and the aqueous phase analysed after a three month period to identify compounds which could potentially be leached from these materials and so contaminate samples. Quantitative results cannot be obtained because the concentrations in solution will depend on the material to water ratio in the test, but potential problem compounds can be identified. Table 4 indicates that significant leaching of 1,1,1 TCE and minor leaching of CCl_3 and CBr_3 is possible. It is encouraging to note that none of the sampled UW piezometers show high levels of 1,1,1 TCE relative to the other volatile organics. This implies that leaching from the piezometer materials has not been significant.

In addition to this quantitative examination of volatile organics, a survey of additional piezometers was carried out in which both halogenated and non-halogenated, volatile organics were examined. Samples were from UW1-4, 3-3, 3-4, 6-2, 7-1, 7-4, 8-2, 9-1, 9-2, 13-2, 14-1, GL9-1 and 11-1 and from the seepage face near UW7. A 2.5 cm^3 sample was purged with purified He at about $30 \text{ cm}^3/\text{min}$ for 11 min. The volatile organics were trapped on sorbant tubes and then thermally desorbed onto a fused quartz capillary column (60 m x 0.32 mm ID, DB-1, SE 30 equivalent) from which the individual components, with some overlaps, were eluted following temperature programming of the column oven (30° for 5 min. up to 150° at $10^\circ/\text{min.}$, then 150° for 5 min.). The components were detected with a halogen-sensitive, electron capture detector or with a hydrocarbon-sensitive flame ionization detector and the results illustrated as response versus elution time in Figures 54 and 55. This system combines a UNACON (Envirochem. Inc.) concentrator and inlet system coupled to a Hewlett-Packard 5840A gas chromatograph.

Figure 54 shows typical gas chromatograms using the electron capture detector. A standard containing about 17 ppb of each of 20 specific, halogenated organics is followed by a system blank (no sample), then by some typical chromatograms of groundwater samples. Note the presence of a contaminant in the analytical system with a retention time of about 16.54 min. An halogenated organic early in the chromatogram is evident in UW7-1 and GL9-1 and three peaks occur at intermediate time for the seepage face. The relative concentrations can be estimated by comparing the peak heights in the samples to those in the standard. It is concluded that no volatile, halogenated organic is

present at concentrations greater than 1 ppb in the sampled groundwaters. This suggests that the very low concentrations listed in the results from Stanford may be representative for this landfill site. This adds credence to the findings of both Mann Testing Laboratory and Stanford that at present these organics are generally absent or in very low concentrations in leachate and the groundwater at the landfill site.

This does not mean that there are no volatile organic compounds present in groundwaters at this site; it only indicates the general lack of halogenated volatile organics. Figure 55 shows gas chromatograms obtained with the flame ionization detector for groundwaters and/or the seepage water. The standard peaks have been attenuated 8 times relative to the blank and samples, so that peak heights in the standard chromatogram should be 8 times larger when comparison is made to peak heights for samples. Correlation of chromatograms can usually be made with the peaks of about 5.60 and 13.53 min. retention times. Groundwaters from locations adjacent to the landfill (UW7-1, 7-4 especially) produce numerous small peaks indicating the presence of numerous volatile organics. The chromatogram of the seepage face waters is very different from actual groundwaters and suggests either a different source or that considerable transformation of organics is occurring in the groundwater. In general, samples removed from the landfill site (UW3-3, 3-4, and 14-1) contain lower concentrations as do some samples adjacent to the site (UW13-2, 9-1, and 9-2 for example). In any case, few (none?) of the volatile organics appear to be present at levels above 1-5 ppb.

These monitoring techniques are relatively simple and inexpensive. They will be further developed and used to delineate the organic volatile contaminants in groundwaters at this site. This will involve gas chromatography/mass spectrometry identification of specific volatile organics with less-expensive gas chromatographic analyses on groundwaters from most piezometers.

Analyses of semi-volatile and non-volatile organics for the groundwaters listed in Table 4 are under way at Stanford University. Results are anticipated before groundwater sampling resumes in April when specific organics will be selected for more detailed monitoring.

Identifiable organics probably make up less than 10% of the dissolved organic matter. The higher-molecular-weight organic matter that makes up the bulk of the TOC is being characterized for a few samples. Of particular interest is the differences between landfill-derived and natural organic matter, and the ability of this organic matter to form stable complexes with metals and trace organic contaminants which would facilitate the migration of these contaminants in groundwaters.

Preliminary metal complexing capacity measurements have been completed on groundwaters from UW9-1 and UW10-1 using a gel filtration technique with a Cu^{2+} - spiked sample. Some iron was found to remain complexed when Cu^{2+} was added and about 0.1 mg/l and 0.3 mg/l Cu^{2+} was organically complexed in UW9-1 and UW10-1 groundwater respectively. This apparent ability of the organics to complex metals and the occurrence of metal concentrations above drinking water criteria in groundwaters raises concern for the possible transport of toxic metals

as stable organic complexes. Additional research is planned in this area.

Preliminary size characterization for the organics in these groundwaters has been carried out using dialysis tubings with nominal molecular weight cut-offs of 2000, and 6000-8000. The 15.2 mg/l TOC in UW9-1 has about 55% less than 2000 MW, 31% between 2000 and 6000-8000 MW and about 14% with a nominal molecular weight greater than about 6000-8000. The distribution of nominal molecular weight sizes for the 6.4 mg/l TOC in sample UW10-1 has about 49% less than 2000 MW, 30% between 2000 and 6000-8000 MW, and about 21% greater than 6000-8000 MW. Most organic contaminants have molecular weights smaller than 2000 and so this fraction is being examined in some detail, especially for groundwaters with higher TOC values.

Environmental Isotope Geochemistry

A preliminary suite of groundwater samples was collected in the fall of 1982, and the concentration of tritium (^3H) and the stable oxygen isotope ratio ($^{18}\text{O}/^{16}\text{O}$) of the water were determined. Results available to date are presented in Table 5. Tritium is a radioactive isotope of hydrogen whose half-life is about 12.4 y. Its measurement is useful in determining the mean time since recharge of the sampled groundwater. Prior to a large rise in tritium in the atmosphere in early 1953 caused by nuclear testing, water that recharged the groundwater zone contained essentially no detectable tritium. Landfilling at the Upper Ottawa Street site began in the early 1950's. Therefore, zones in which landfill-derived contaminants occur should also be zones in which tritium occurs.

Present day groundwater recharge in southern Ontario contains about 50 to 100 Tritium Units (T.U.). Groundwater recharge between 1960 and 1970 contained much higher levels (200-2000 T.U.) due to large atmospheric nuclear tests. The tritium content of groundwaters from the Upper Ottawa Street site ranges from 120 T.U. to essentially 0 T.U. This indicates that the bulk of these groundwaters have been recharged since 1953. At the few locations where the tritium values are low, the groundwater may be much older. Because of possible contamination by drilling fluid and piezometer leakage, another series of samples must be collected before a final interpretation of the tritium distribution is made. Similarly, the $^{18}\text{O}/^{16}\text{O}$ ratio measurements in groundwaters must be completed and repeated but there is an indication that water masses of different isotopic compositions are present. Many groundwater samples have $\delta^{18}\text{O}$ values in the range from $-9.6^0/_{\text{oo}}$ to $-10.2^0/_{\text{oo}}$ which could represent groundwaters uncontaminated by drilling fluid. This could indicate a range of environmental conditions (temperature) during recharge. Less negative values could represent unusual water masses or drilling fluid contamination. Temporally variable results may indicate removal of this contamination or the presence of waters recharged under different climatic conditions. Isotopic studies at other landfill sites suggest waters emanating from landfills may have altered isotopic ratios distinct from uncontaminated waters. More extensive sampling is required to fully utilize this isotope data.

SUMMARY AND CONCLUSIONS

The Upper Ottawa Street landfill site is underlain by a number of shale, limestone and dolostone units, all of which are horizontally bedded. Fractures are evident along bedding planes. Vertical or near vertical fractures also exist in these strata.

Nineteen multilevel monitoring devices were installed in bedrock at the site during four episodes of drilling that occurred in April, May, July and October, 1982. A single multilevel device was installed in 1981. Each multilevel device was installed in a borehole from which continuous or nearly continuous core samples were obtained. Inspection of the cores and the results of water-level rise tests in the piezometers in the multilevel devices indicate that open fractures cause all of the rock units to have some degree of hydraulic conductivity. Gross values of horizontal hydraulic conductivity range from very low values, on the order of 10^{-10} cm/s, to high values, on the order of 10^{-3} cm/s.

Water-table monitoring in the multilevel devices indicates that the general water-table slope is from west to east from the landfill towards Albion Falls and the Niagara Escarpment. A water table mound exists within or beneath the landfill. Lateral flow of leachate occurs radially outward from the landfill. Lower hydraulic-head levels exist in deeper zones in the vicinity of the landfill. This indicates that there is some potential for downward movement of leachate

from shallow active groundwater flow zones into deeper zones. Downward components of the hydraulic gradient are generally much larger than the lateral components of the hydraulic gradient, but the horizontal hydraulic conductivity is probably much higher than the vertical hydraulic conductivity. As a result, it is expected that the bulk of the contaminated groundwater will move laterally. It is not possible at present to determine the relative proportions of leachate migration between the lateral and vertical pathways.

It was necessary to use water obtained from local hydrants for circulation in the boreholes during drilling. Because many of the groundwater monitoring zones have low permeability and therefore yield water at only very slow rates when piezometers are pumped, many weeks or months of repetitive episodes of pumping are necessary to completely remove the drilling water from the monitoring zones.

Most of the piezometers in the network of multilevel devices were sampled on two or more occasions. These samples were analysed for electrical conductance, chloride and total dissolved organic carbon. Based on these parameters it has been possible to recognize groundwater zones that are grossly contaminated by chloride and organic carbon from the landfill. Zones that are not grossly contaminated but which may be mildly or only slightly contaminated may not be distinguishable using these routine parameters. This is the case because the natural groundwater in some zones in the rock, particularly in the shale, contain chloride and organic carbon contents that are exceptionally high relative to what is normally encountered in natural groundwater in other types of hydrogeologic systems.

For definition of the extent and nature of the zone of leachate contamination, groundwater samples for analysis of many other constituents were collected. Many of the groundwaters adjacent to the landfill and to the east have concentrations of NH_3 , B, Cl, Cr, Fe, Pb, Mn and SO_4 in excess of drinking water criteria. The distribution of these possible contaminants is not regular and the possible variations in background water quality complicates the definition of a leachate-contaminated plume based on inorganic analyses. Methane is produced within the landfill and also occurs in many of these groundwaters. Before the distribution of methane can be used to delineate landfill-leachate contamination, carbon isotope studies must be undertaken to help differentiate landfill-derived methane from methane which occurs naturally in the bedrock.

The general lack of volatile, halogenated organics in groundwaters at this site was surprising and their usefulness in delineating landfill-leachate contamination is questionable. Numerous, volatile, but not halogenated organics are present in these groundwaters. The studies at Stanford University should identify many of these and it is hoped that some may have a unique source (i.e. the landfill) and so may be used to document the extent of landfill contamination of groundwaters.

Environmental isotope analyses confirm the variable nature of groundwater obtained over time from some piezometers. Continued monitoring is required to document the complete removal of drilling fluids especially from zones of low permeability. On the other hand, the sampling protocol development study suggests that waters of variable

chemistry will be obtained during the pumping from piezometers and this could reflect the presence of waters of variable chemical (and isotopic?) composition in this low-storage material.

At this stage the extent and detailed geochemical nature of the zone of leachate contamination are not well defined. It is evident that detailed consideration of the hydrogeology, inorganic, organic and isotope geochemistry will be required to confidently address this problem. All aspects will be vigorously addressed in 1983/84 and the integration of the various methods should provide a firm scientific basis for delineation of groundwater contamination at the Upper Ottawa Street landfill site.

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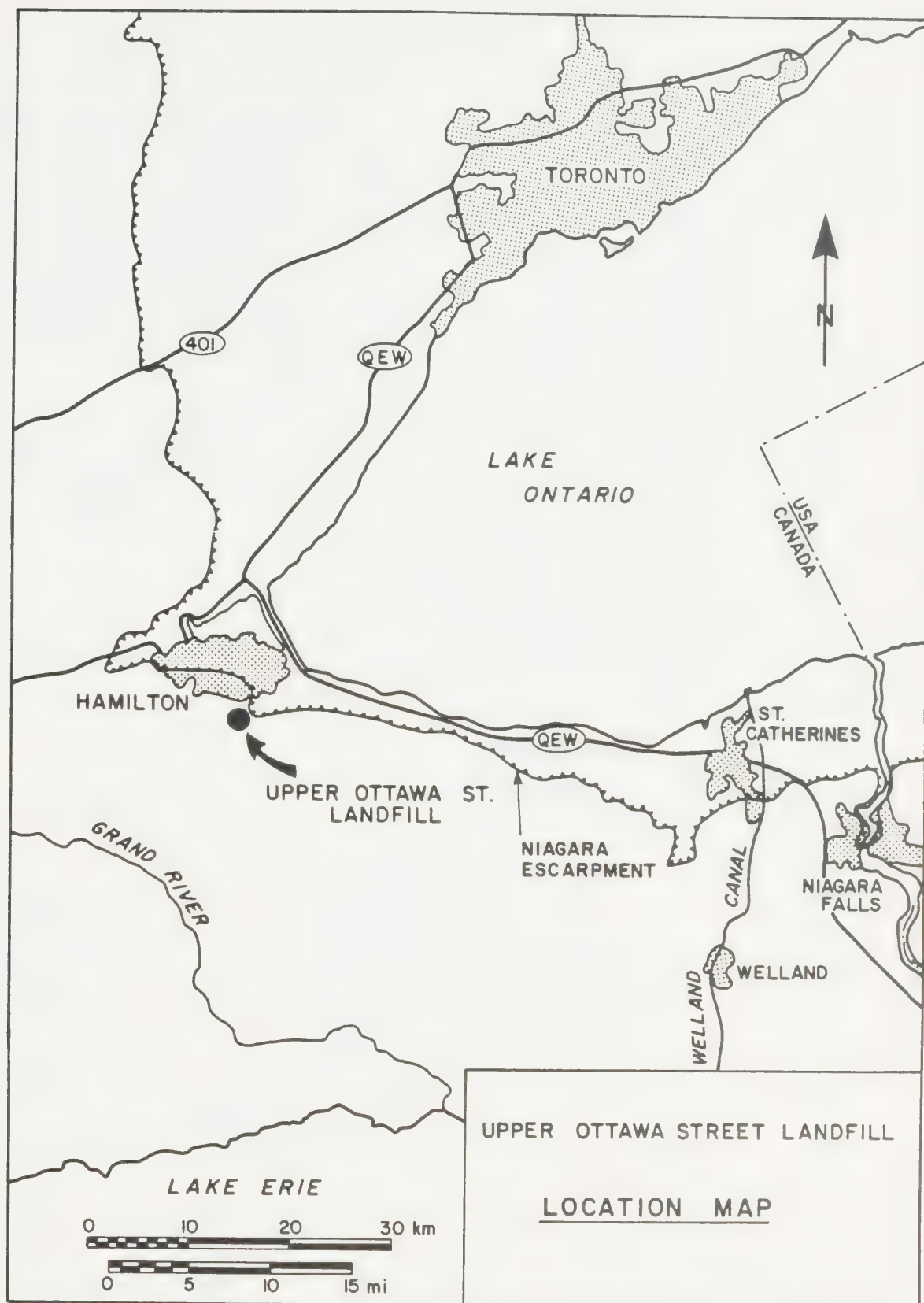


Figure 1. Location of the Upper Ottawa Street Landfill

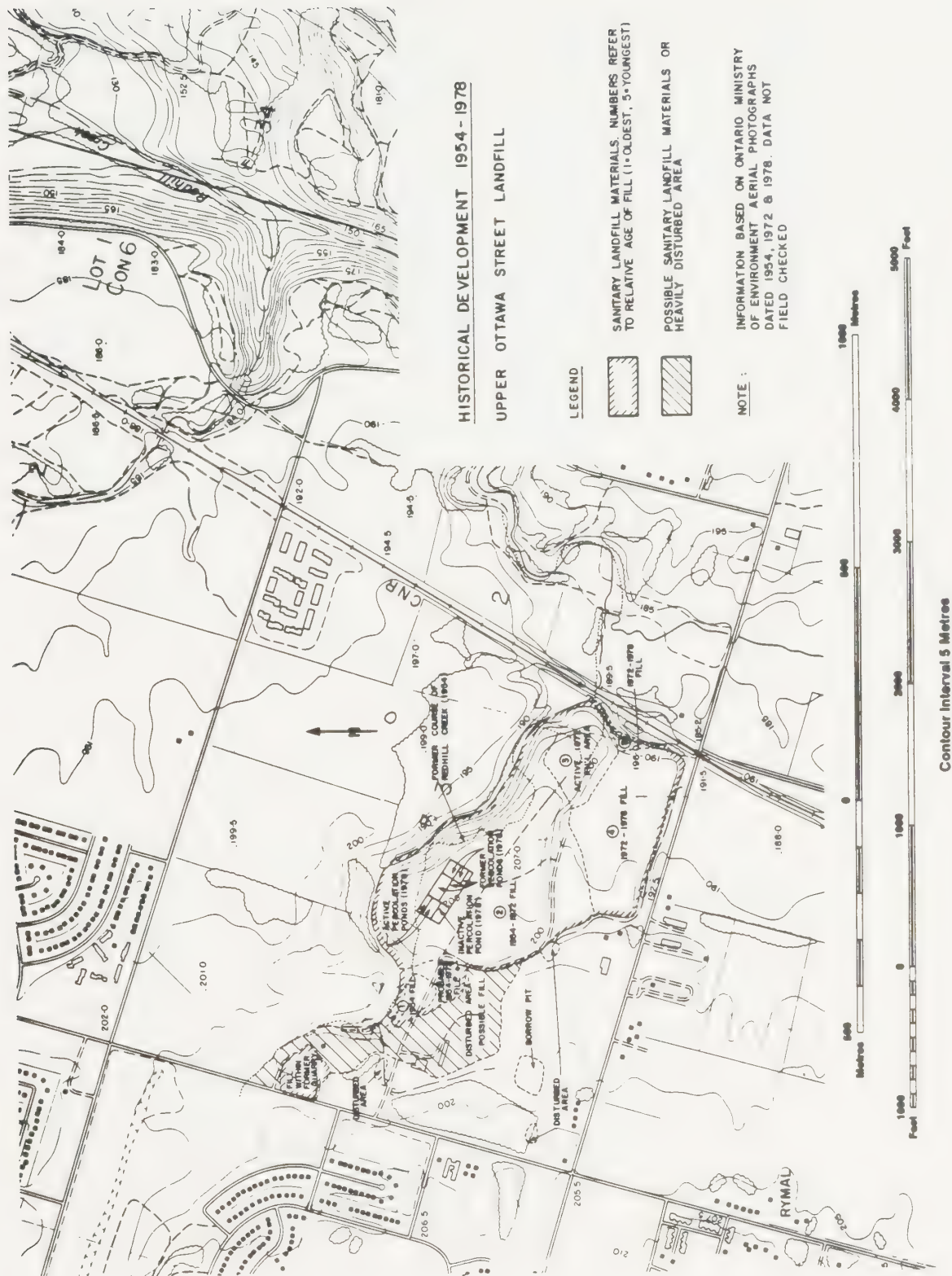


Figure 2. Historical Development

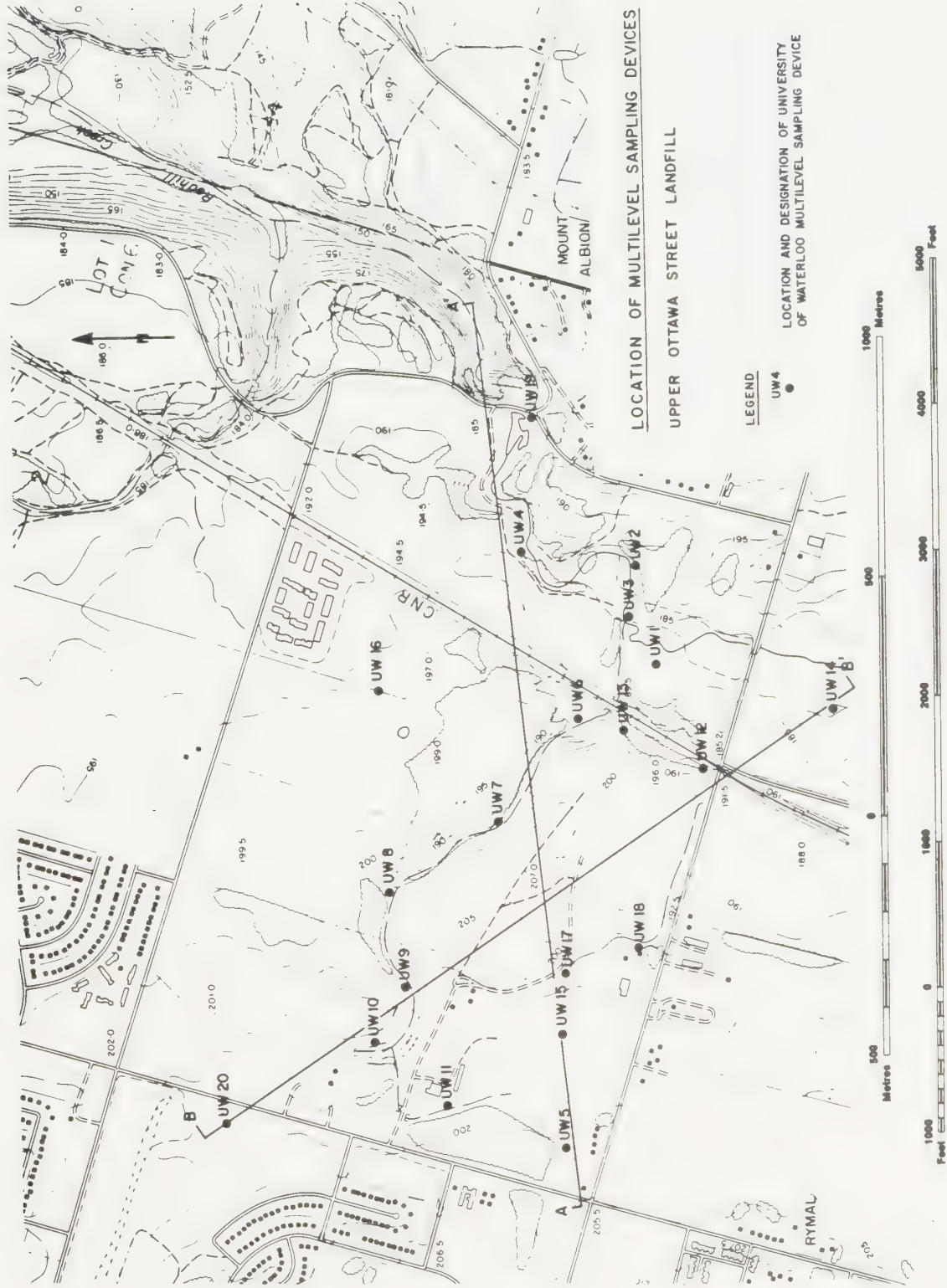


Figure 3. Location of University of Waterloo
Multilevel Sampling Devices

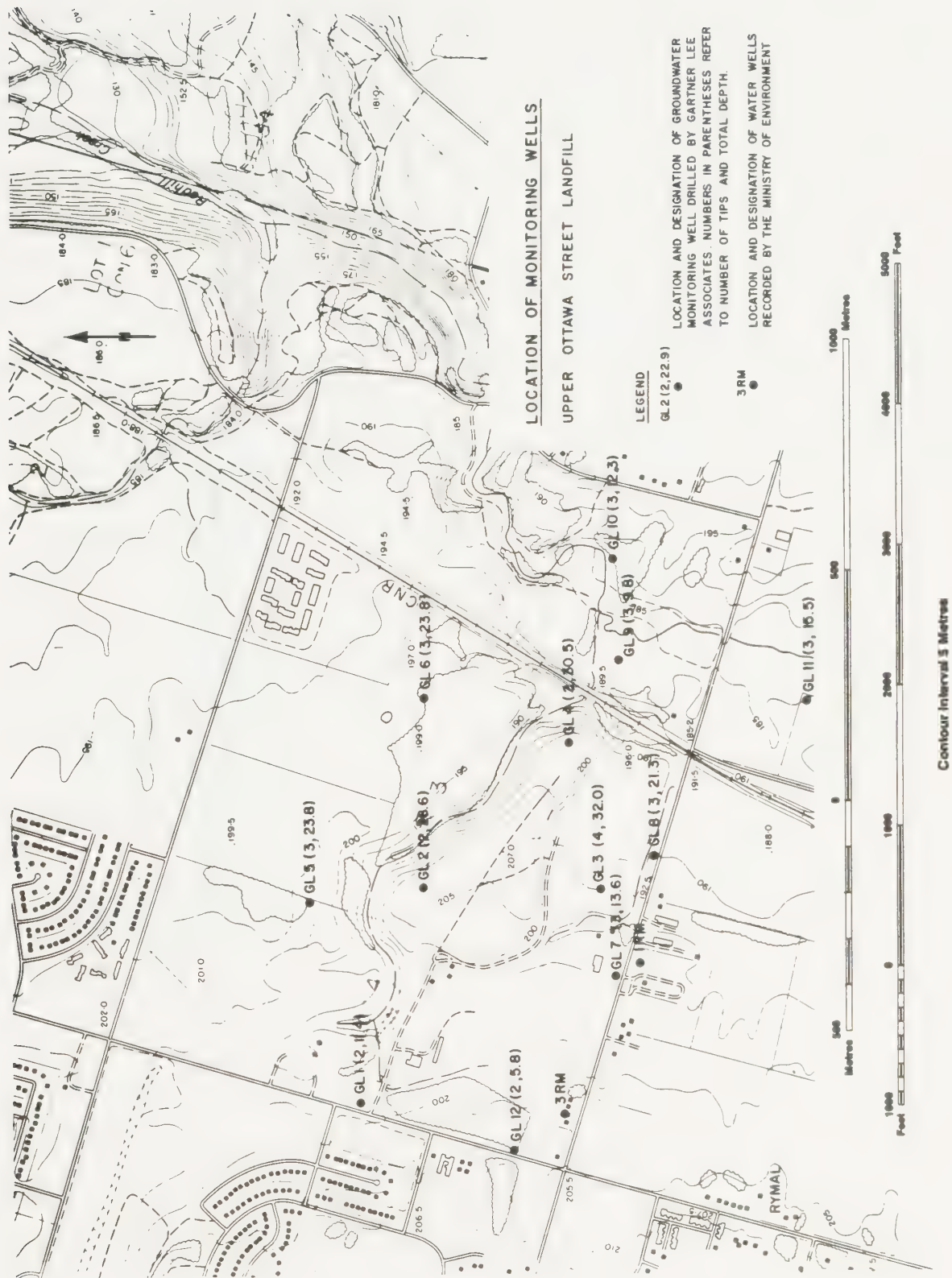


Figure 4. Location of Monitoring Wells existing prior to University of Waterloo study

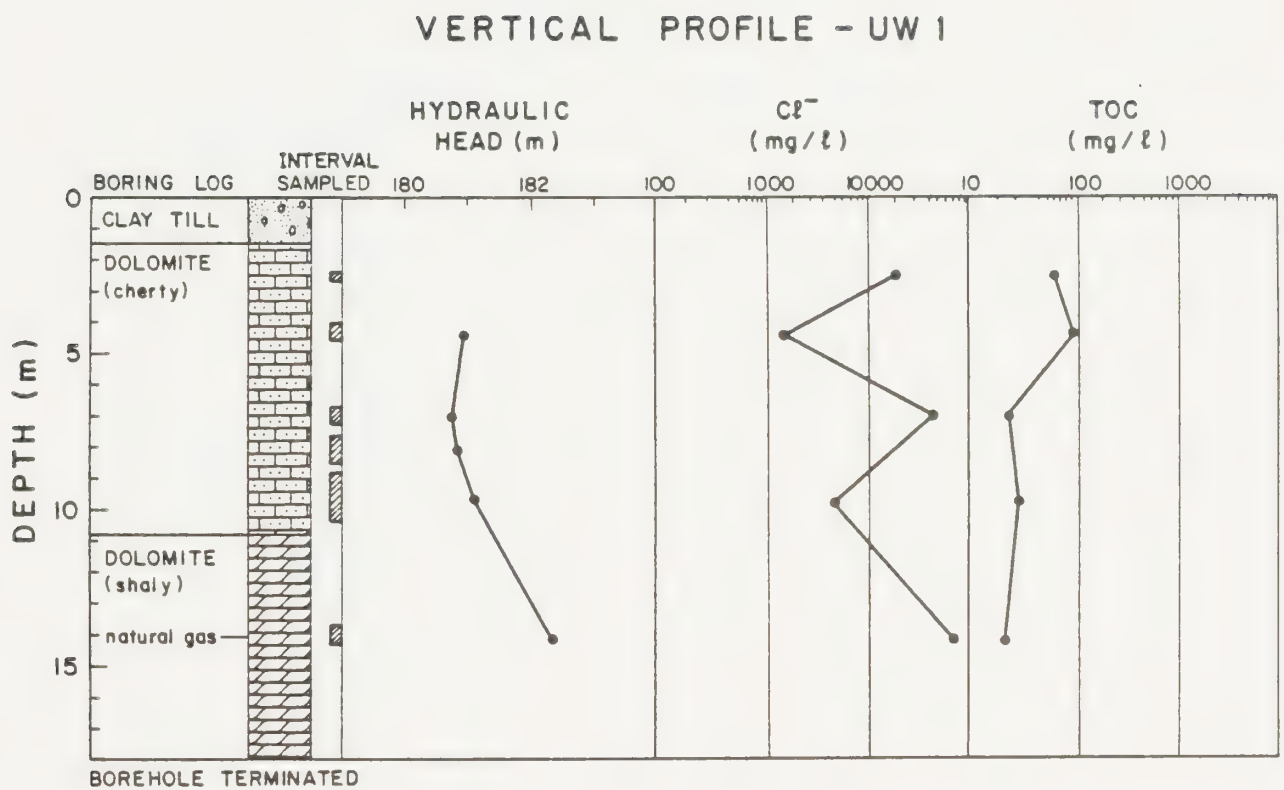


Figure 5. Vertical Profile - UW 1

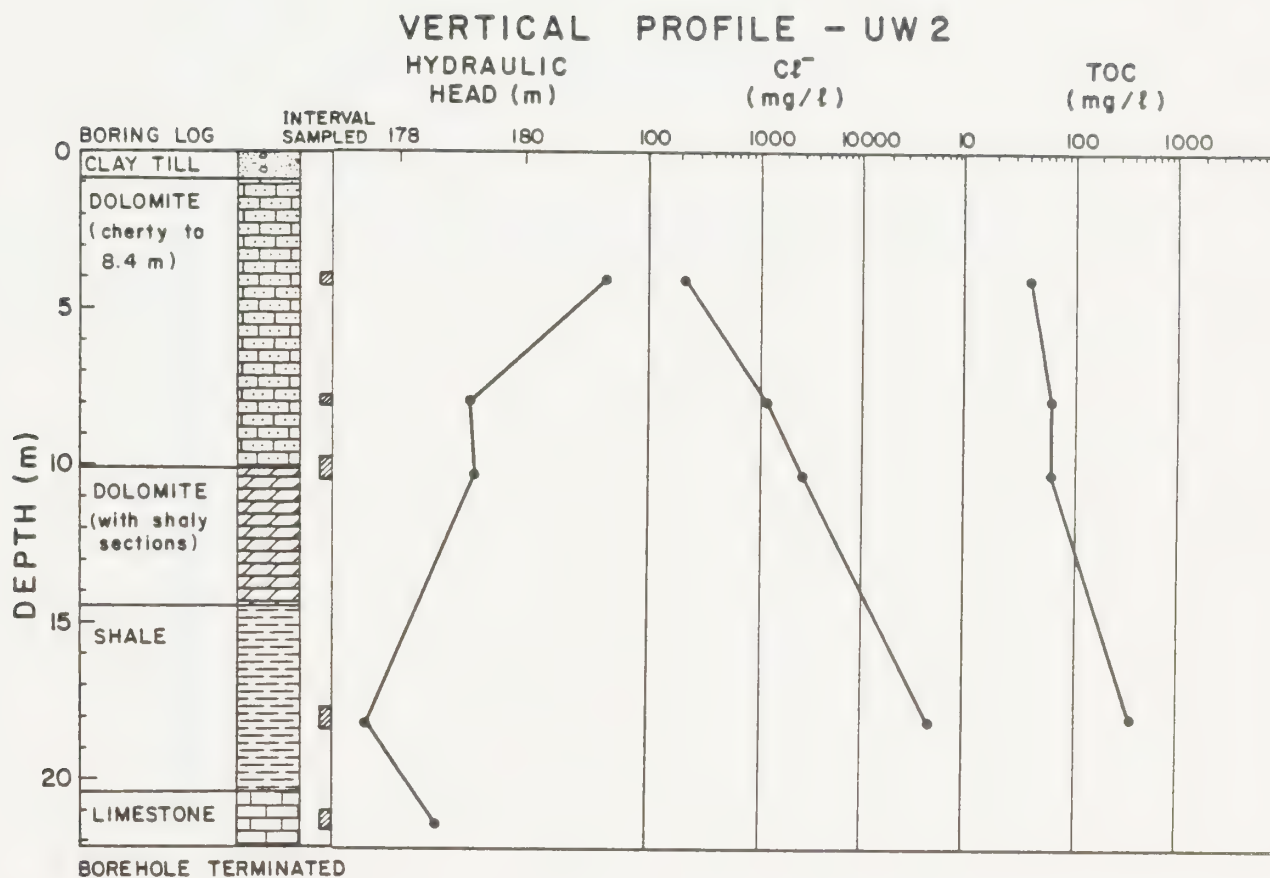


Figure 6. Vertical Profile - UW 2

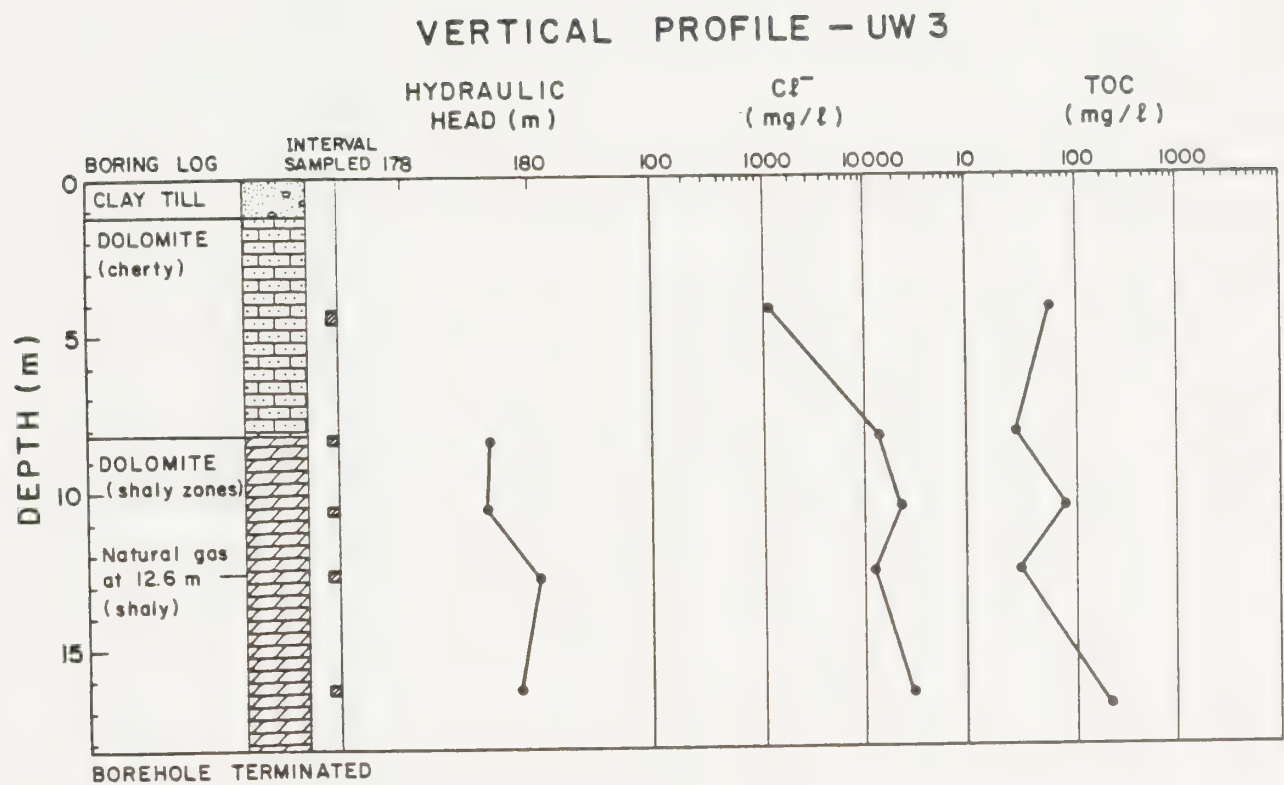


Figure 7. Vertical Profile - UW 3

VERTICAL PROFILE - UW 4

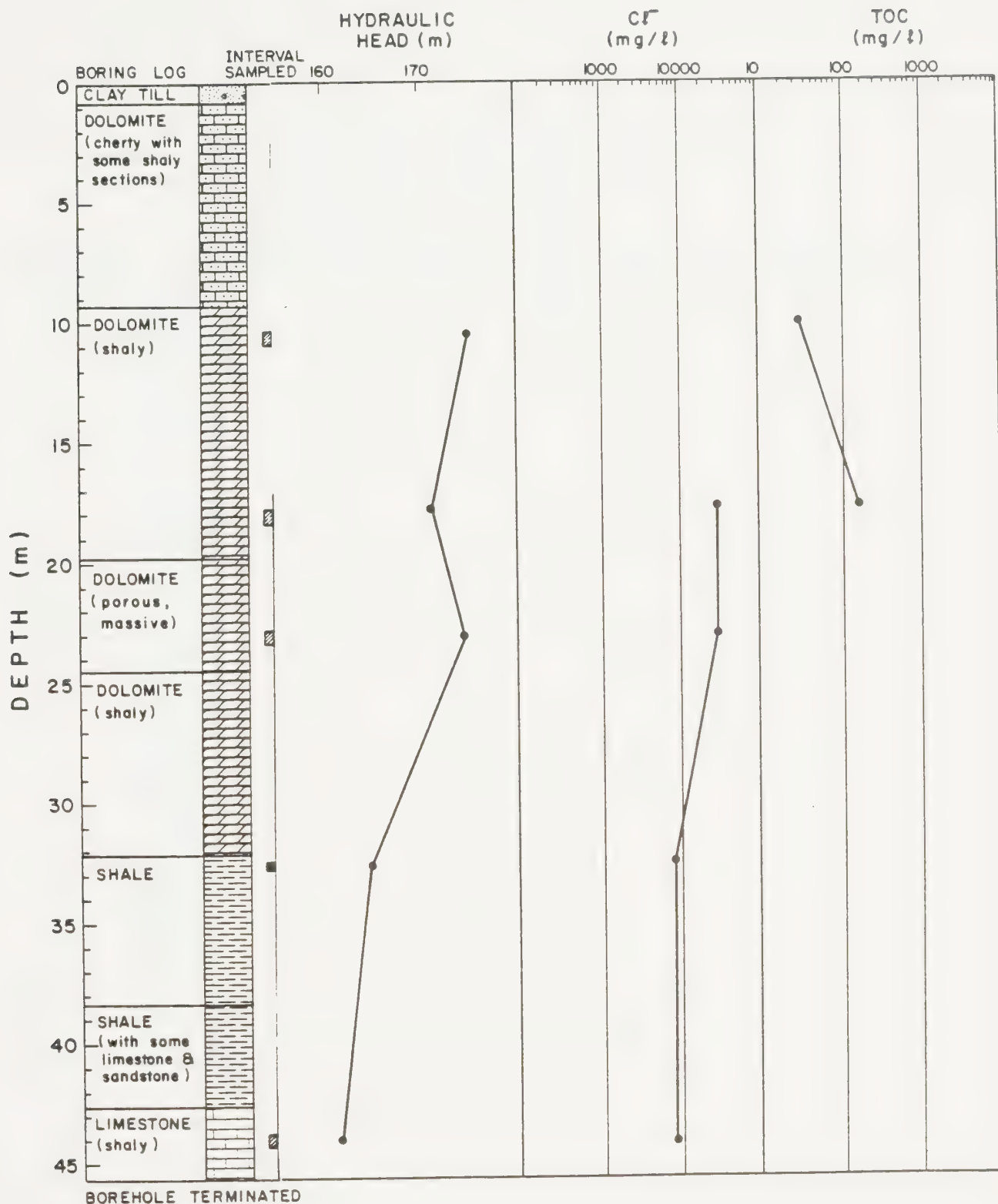


Figure 8. Vertical Profile - UW 4

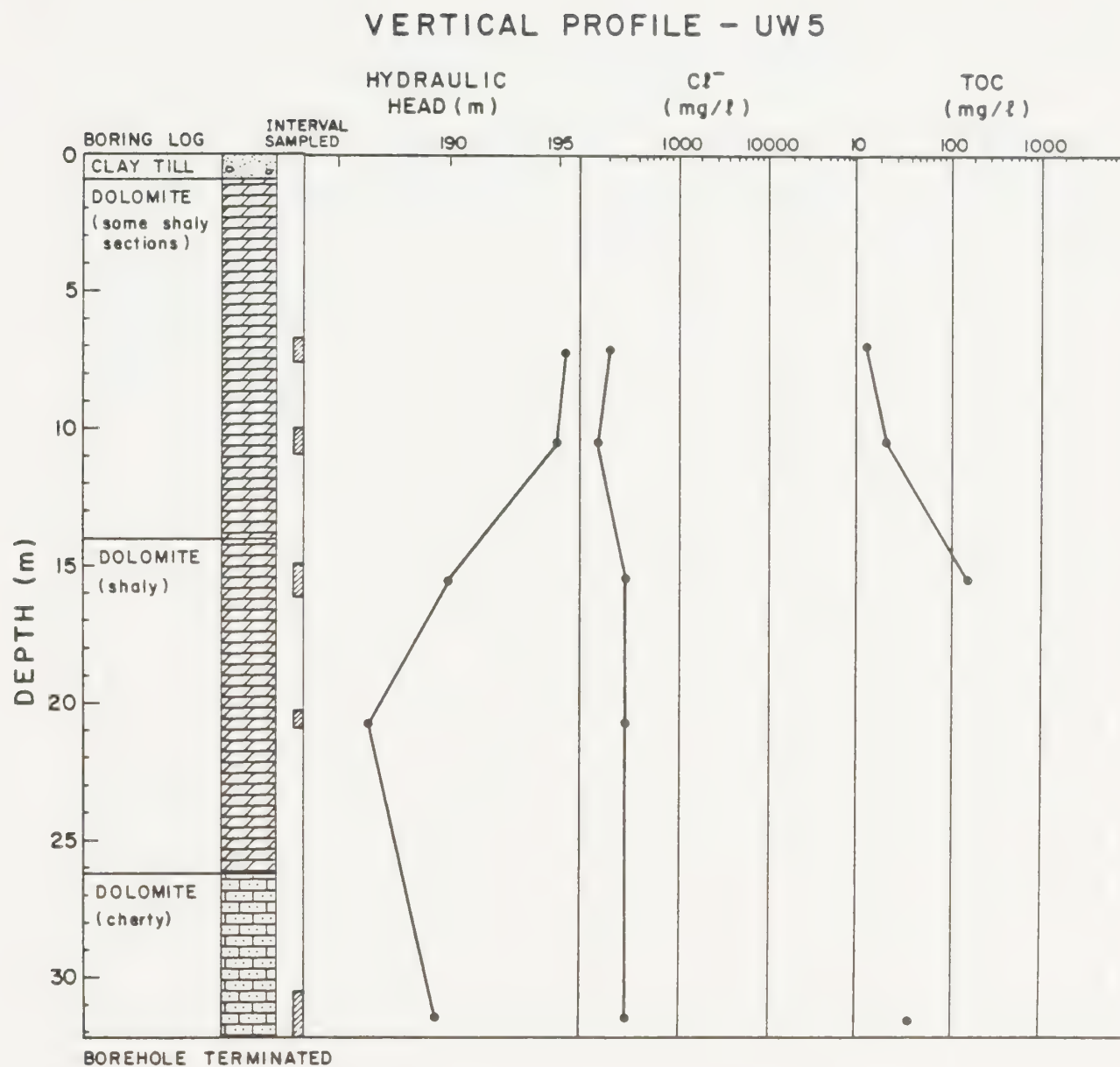


Figure 9. Vertical Profile - UW 5

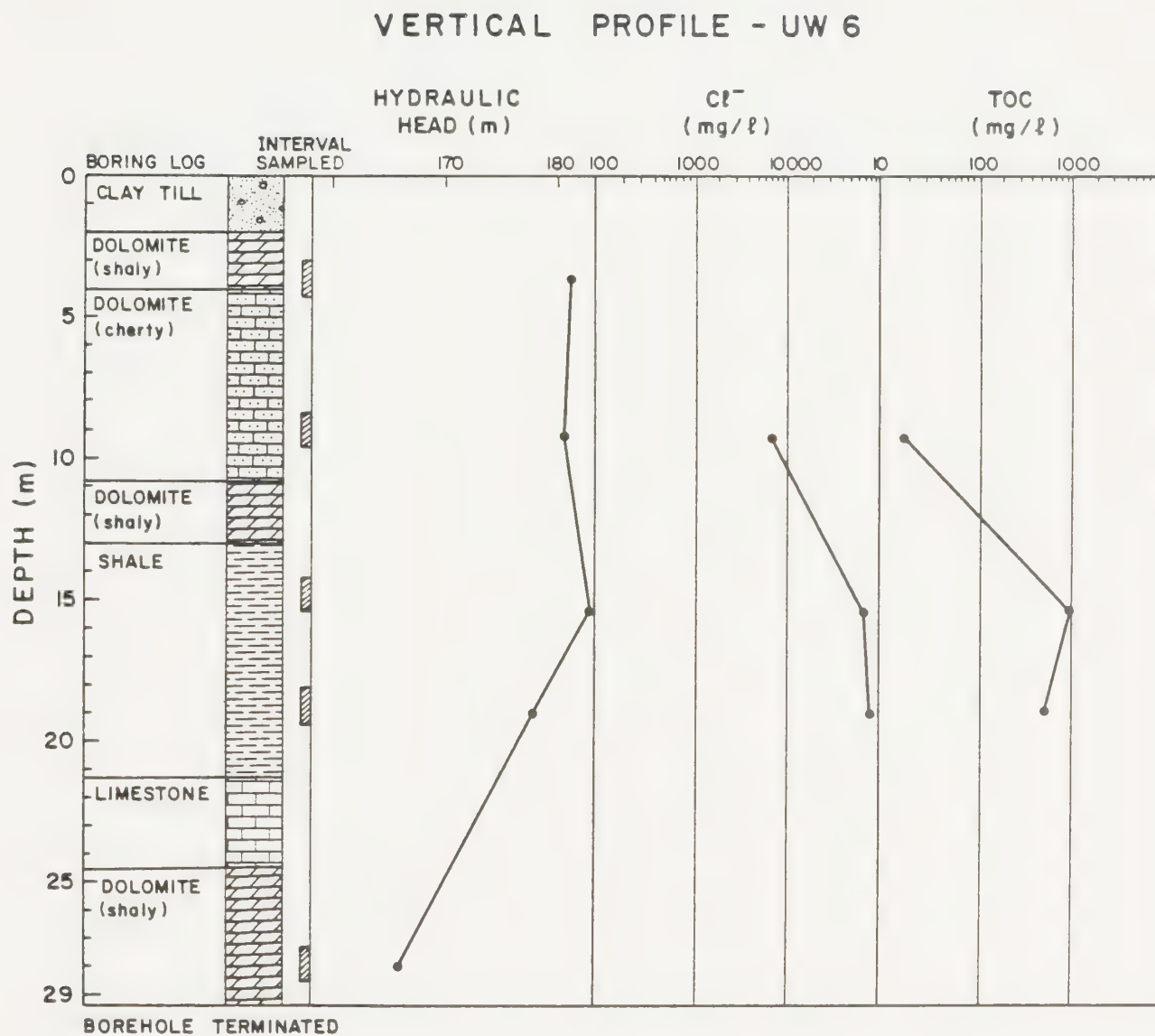


Figure 10. Vertical Profile - UW 6

VERTICAL PROFILE - UW 7

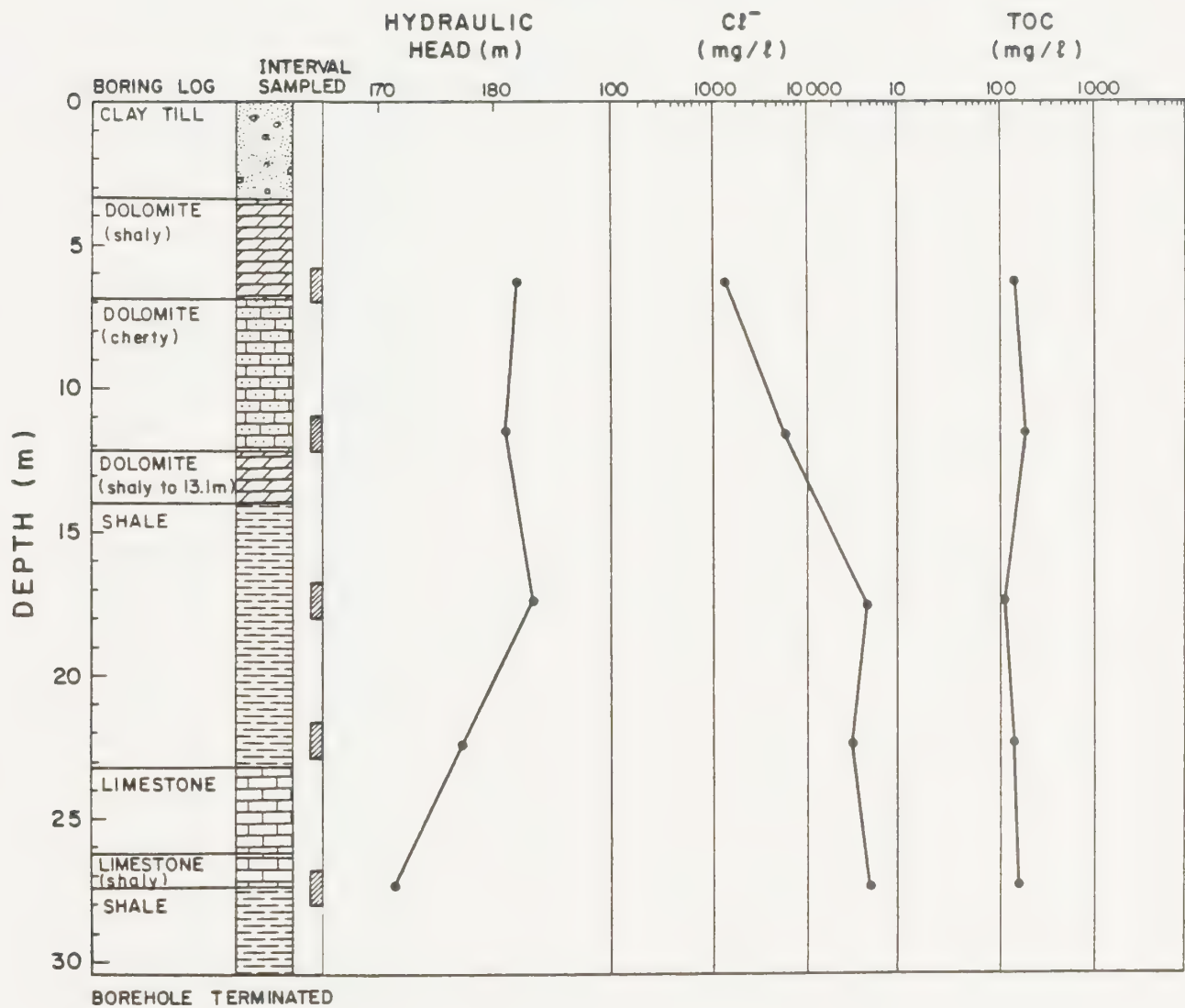


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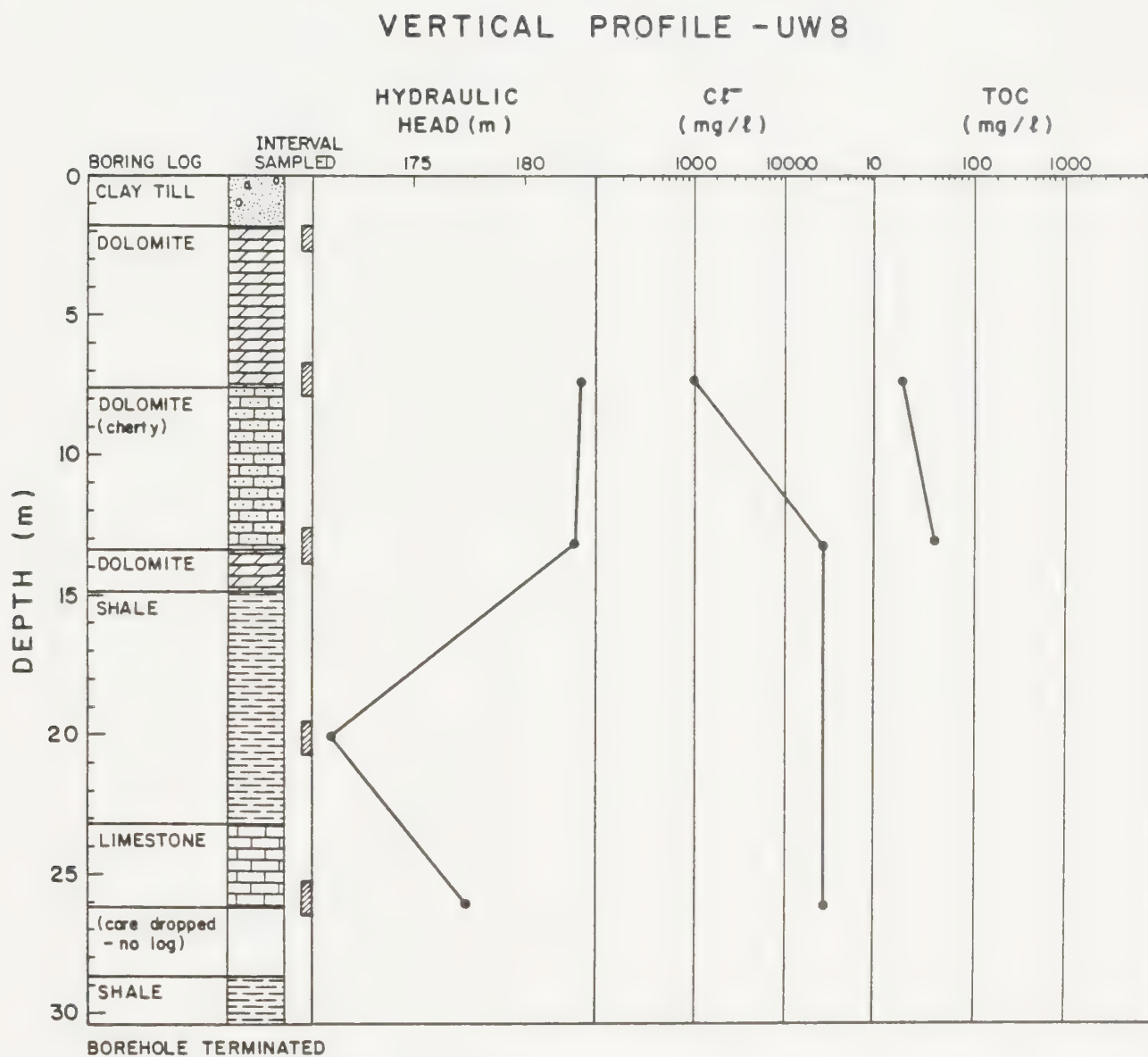


Figure 12. Vertical Profile - UW 8

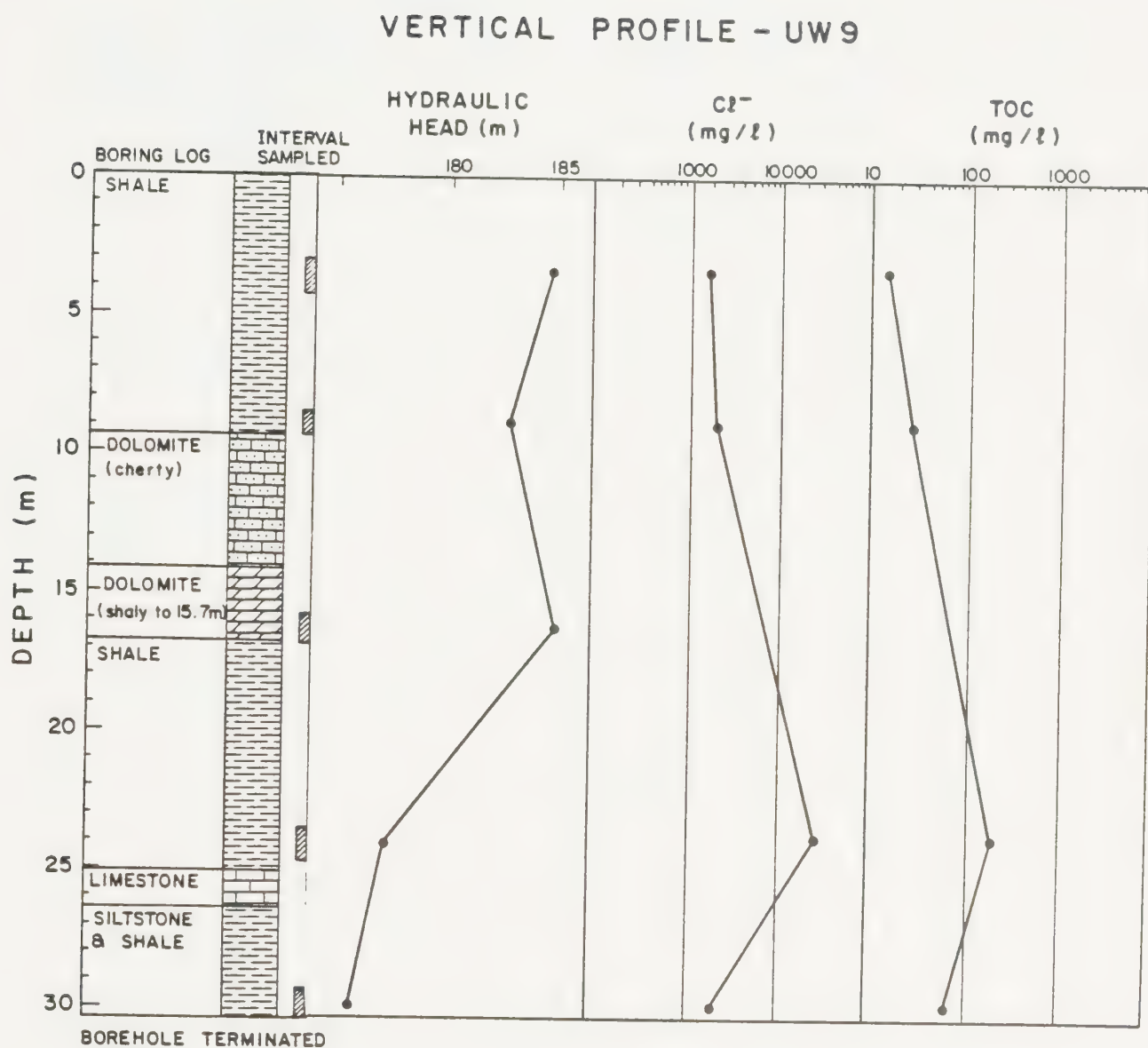


Figure 13. Vertical Profile - UW 9

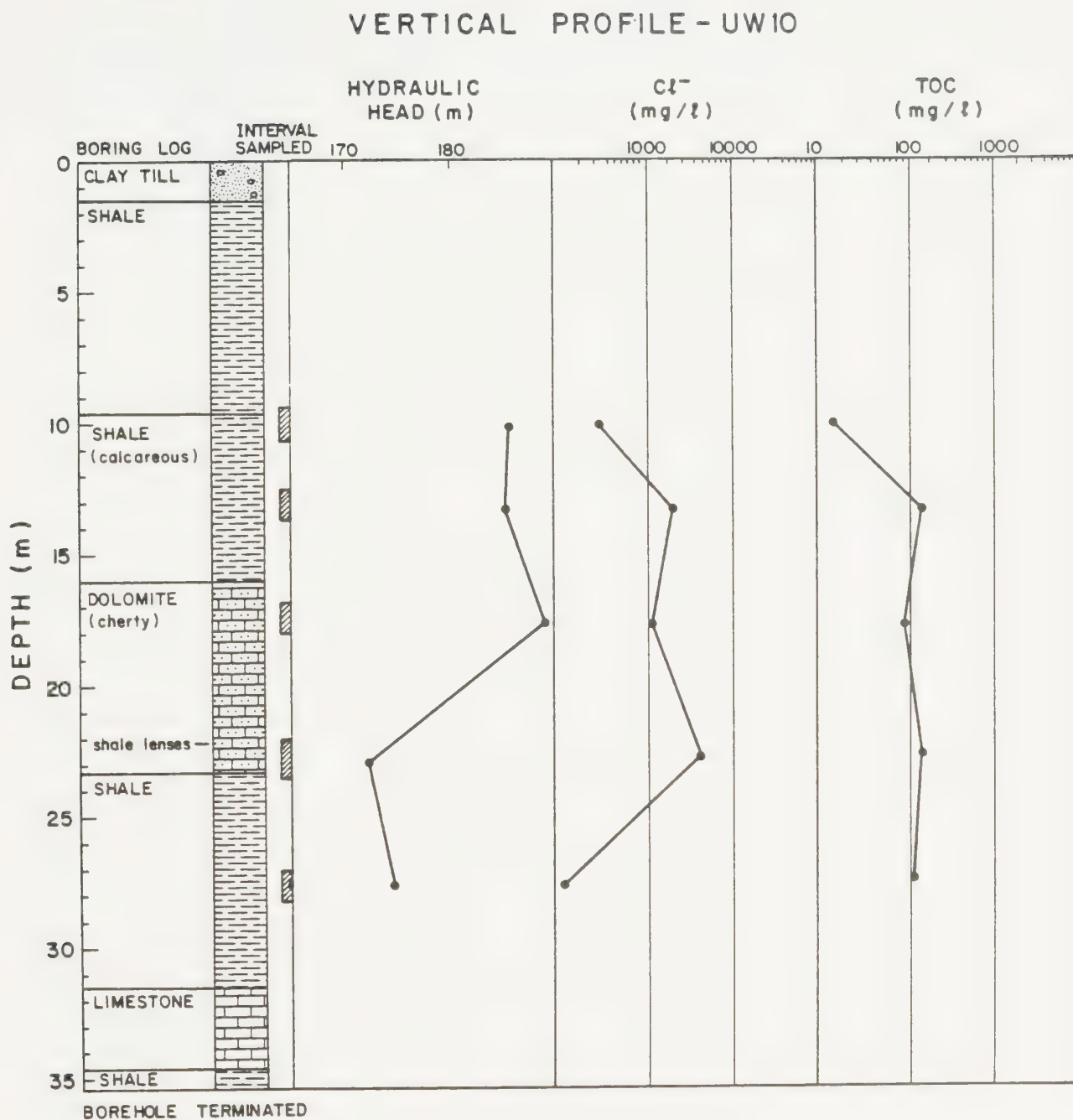


Figure 14. Vertical Profile - UW10

VERTICAL PROFILE - UW 11

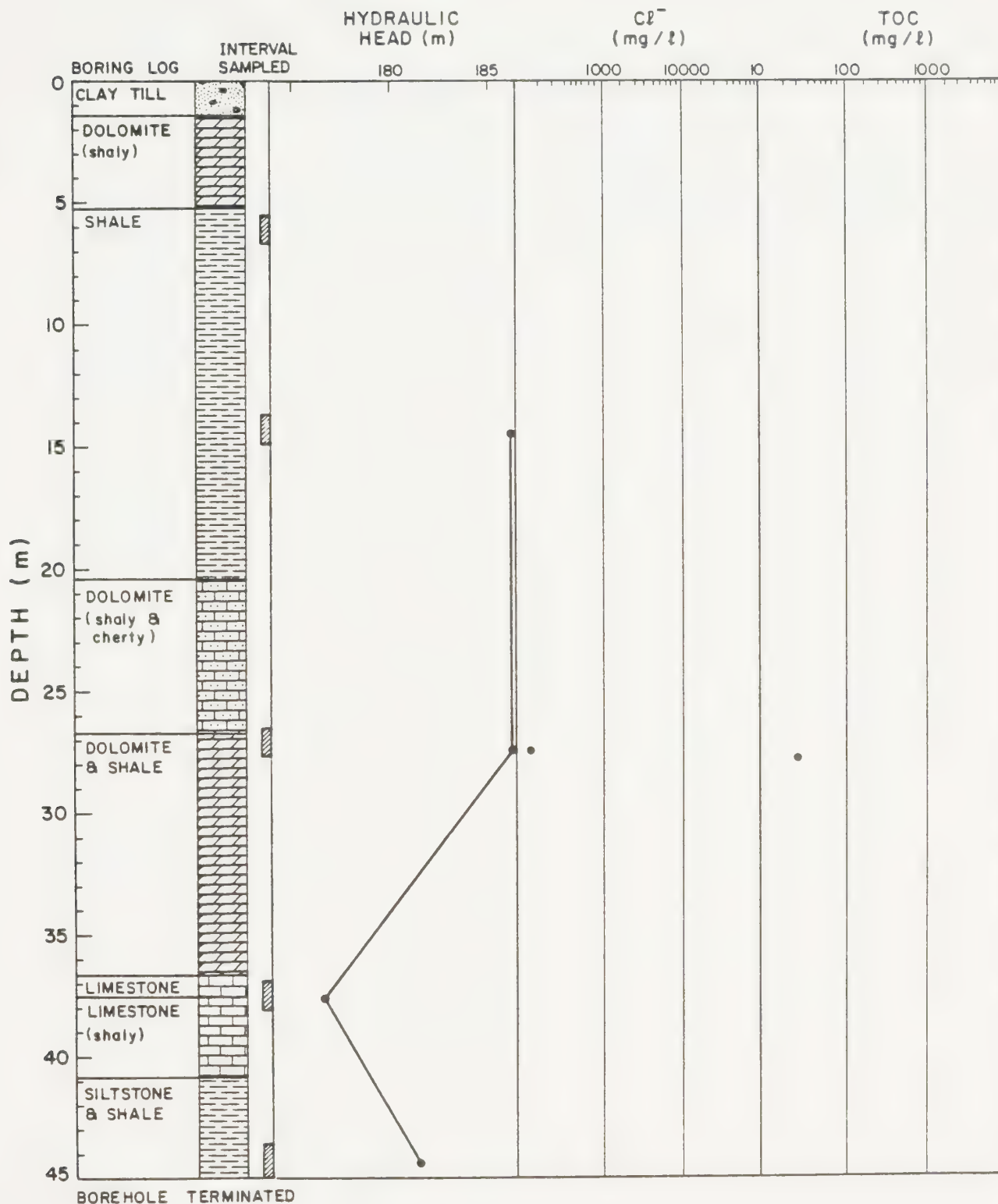


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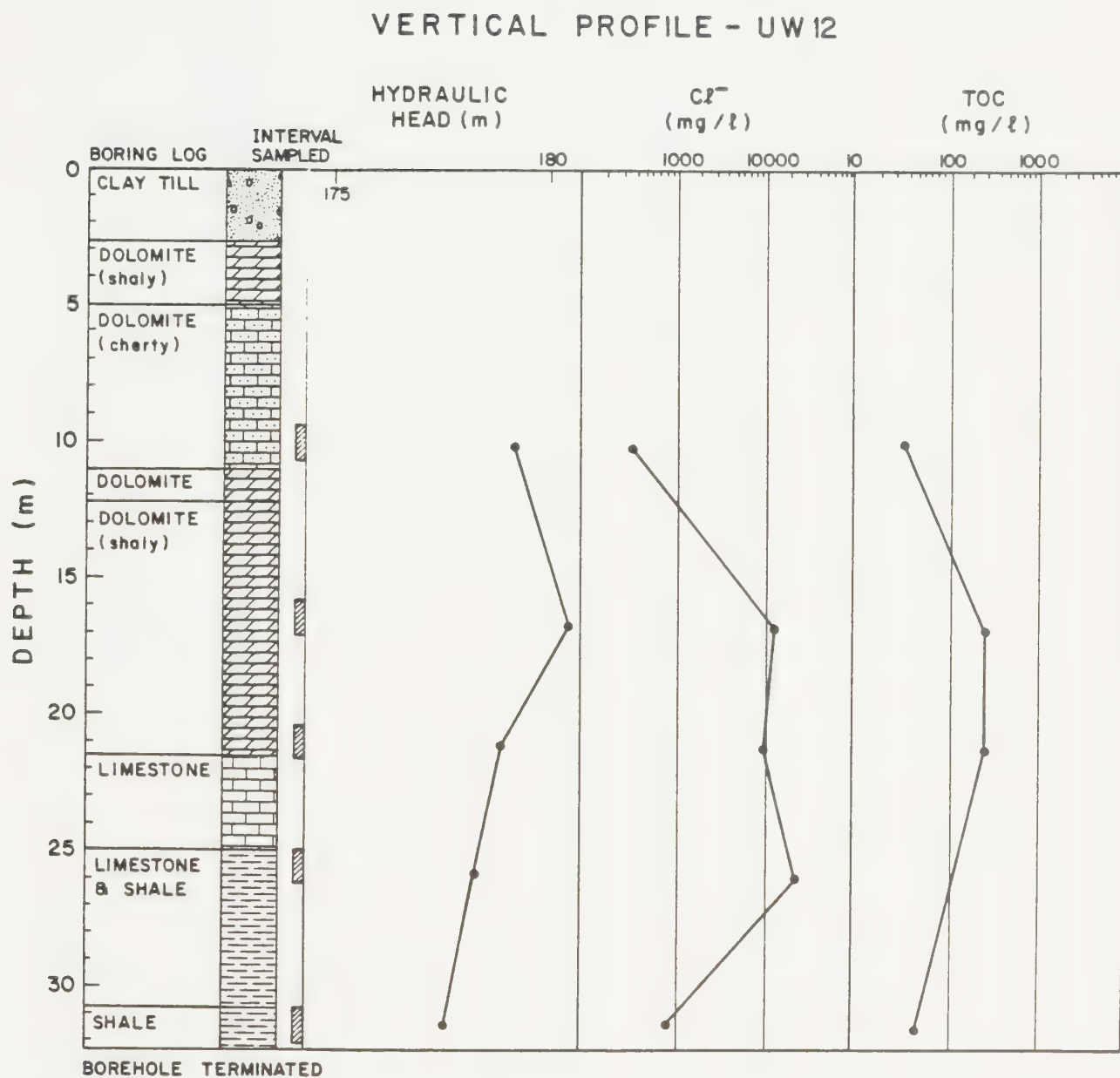


Figure 16. Vertical Profile - UW12

VERTICAL PROFILE - UW 13

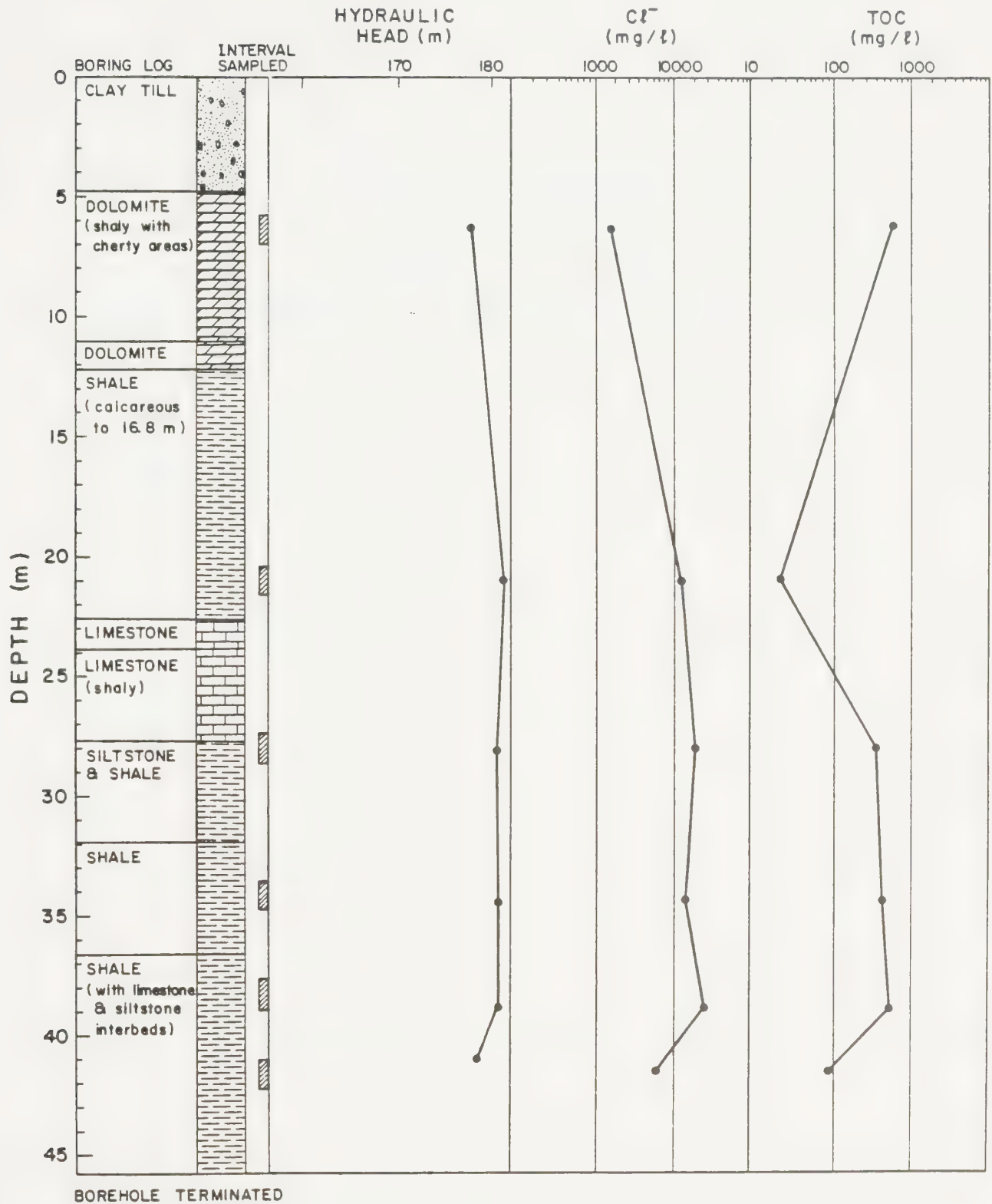


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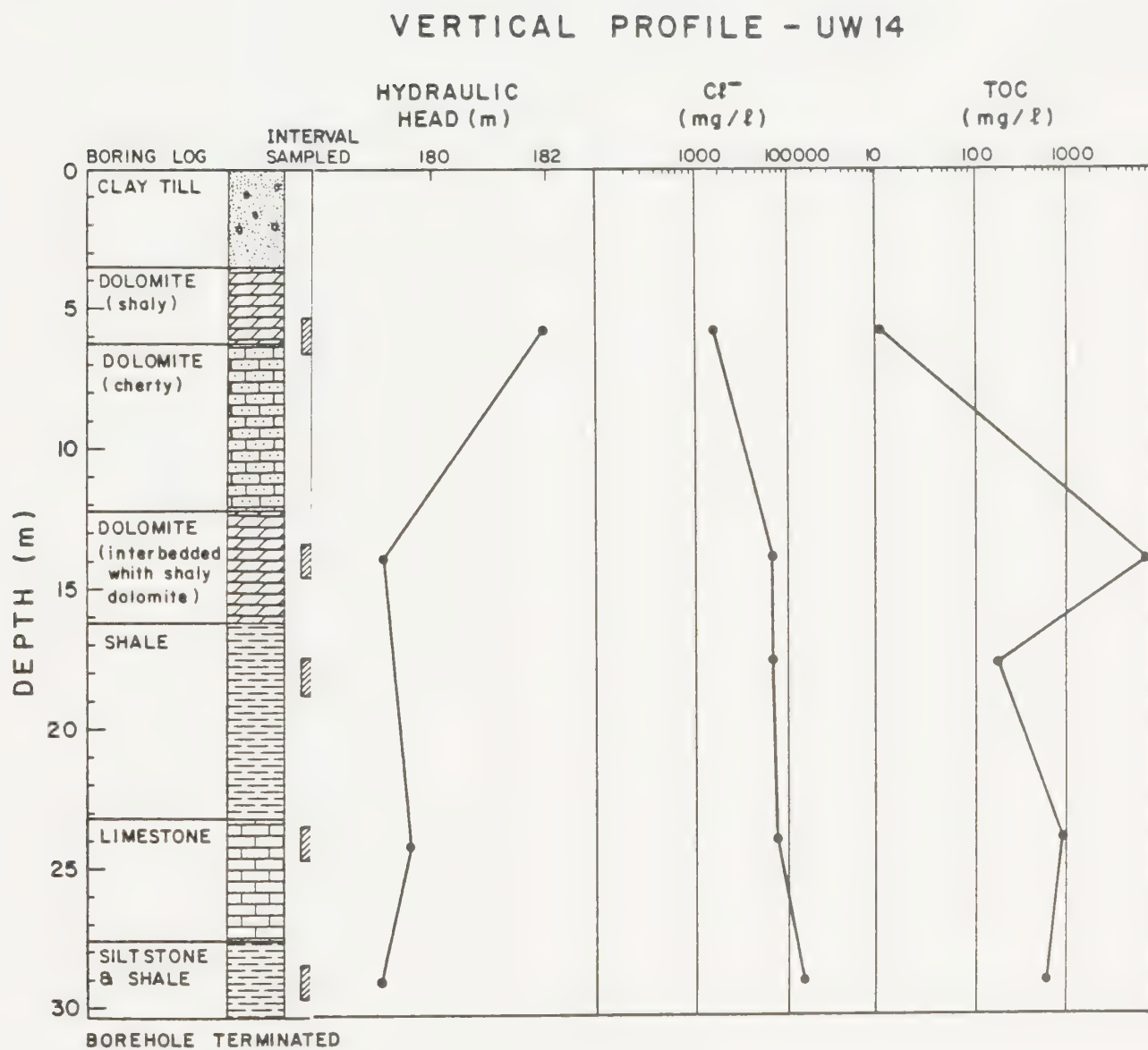


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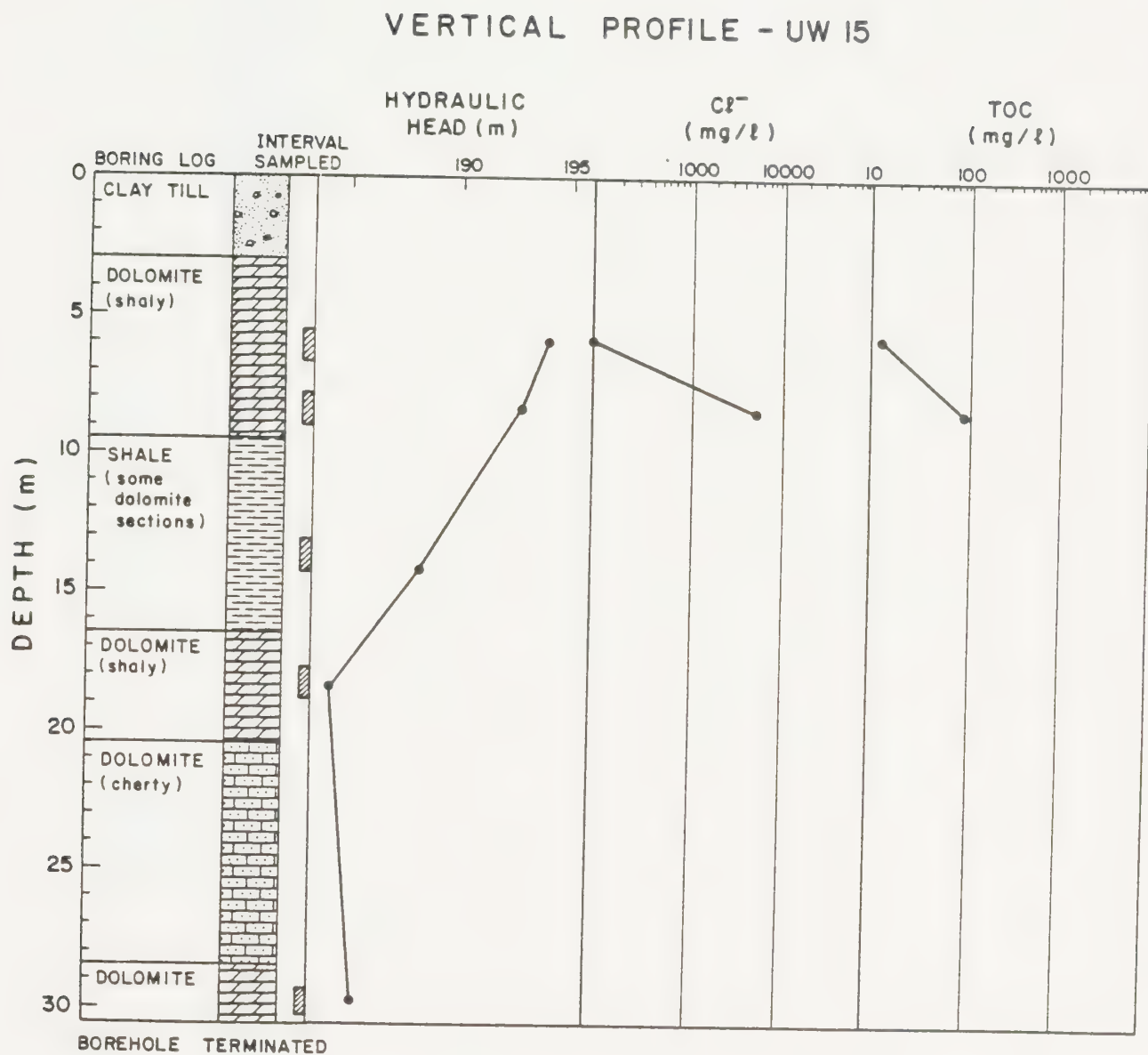


Figure 19. Vertical Profile - UW15

VERTICAL PROFILE - UW 16

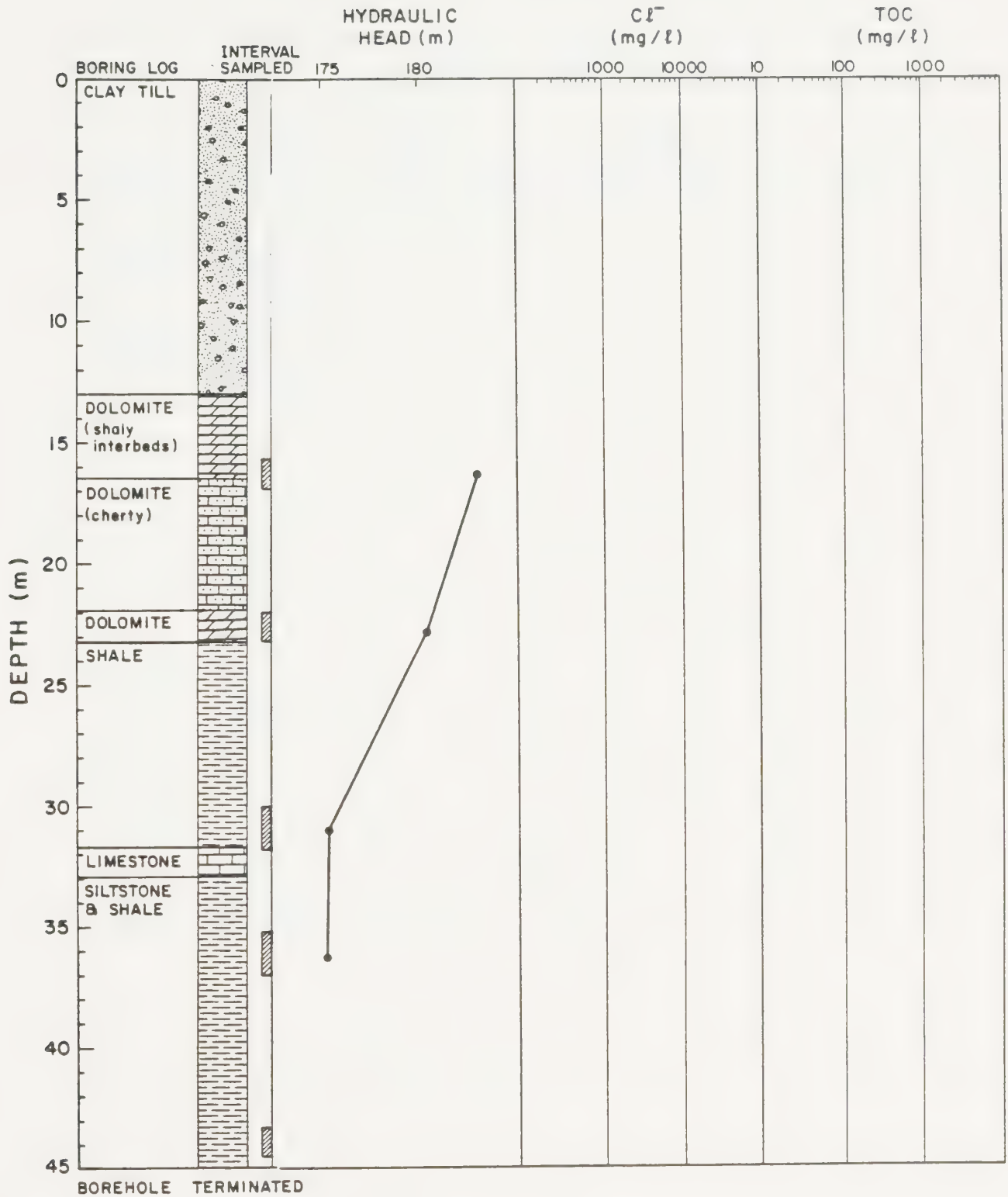


Figure 20. Vertical Profile - UW16

VERTICAL PROFILE - UW 17

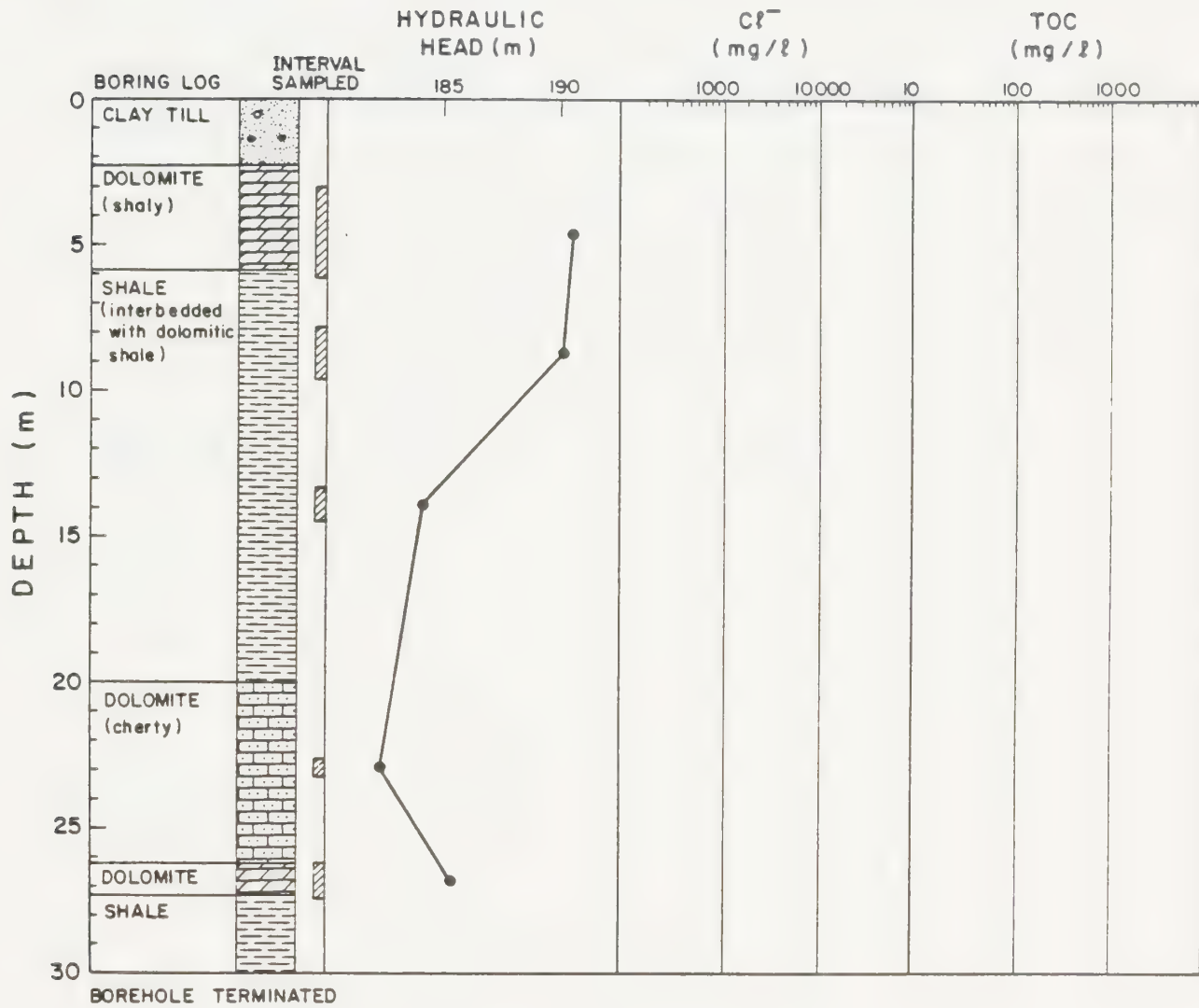


Figure 21. Vertical Profile - UW17

VERTICAL PROFILE - UW 18

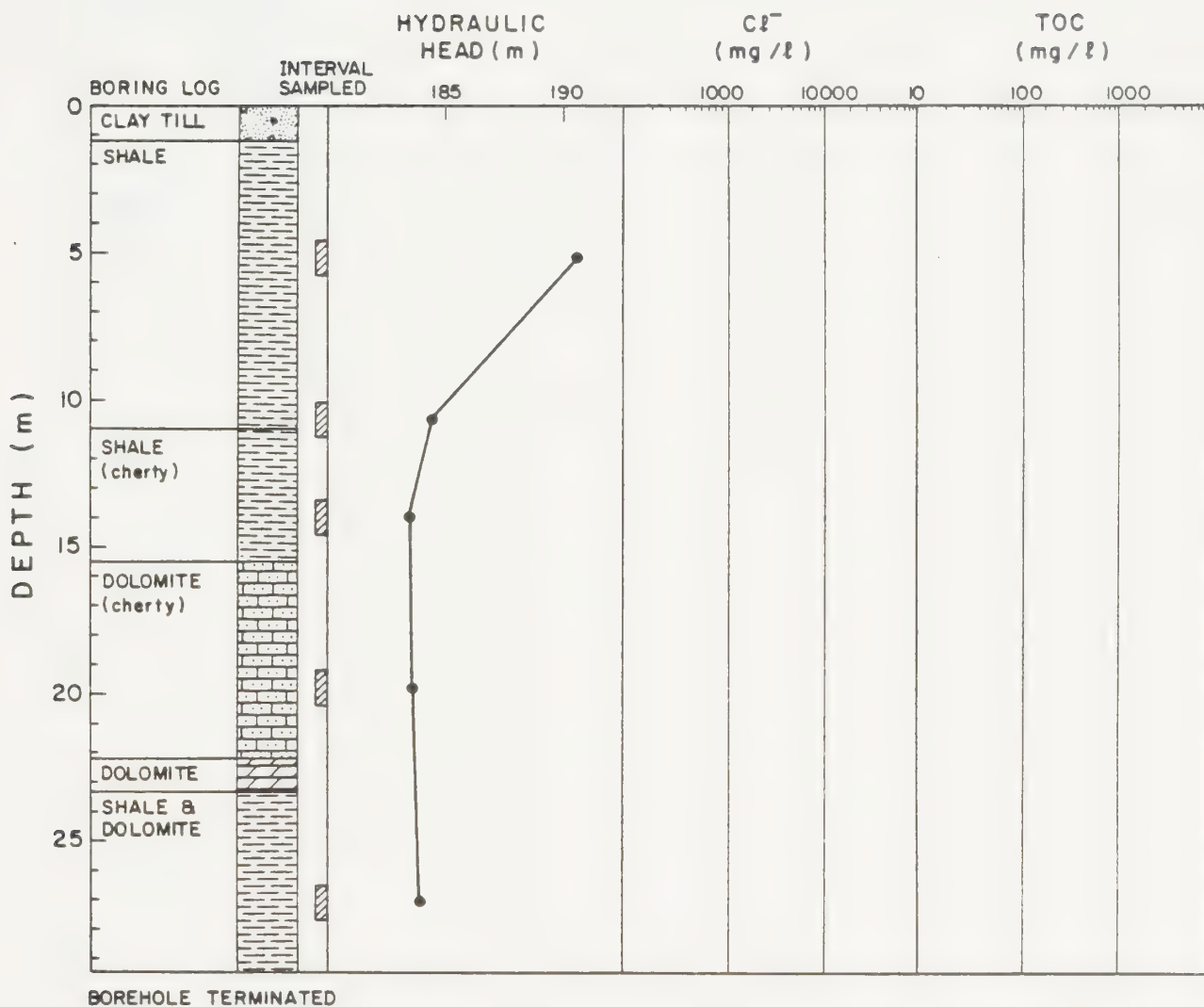


Figure 22. Vertical Profile - UW18

VERTICAL PROFILE - UW 19

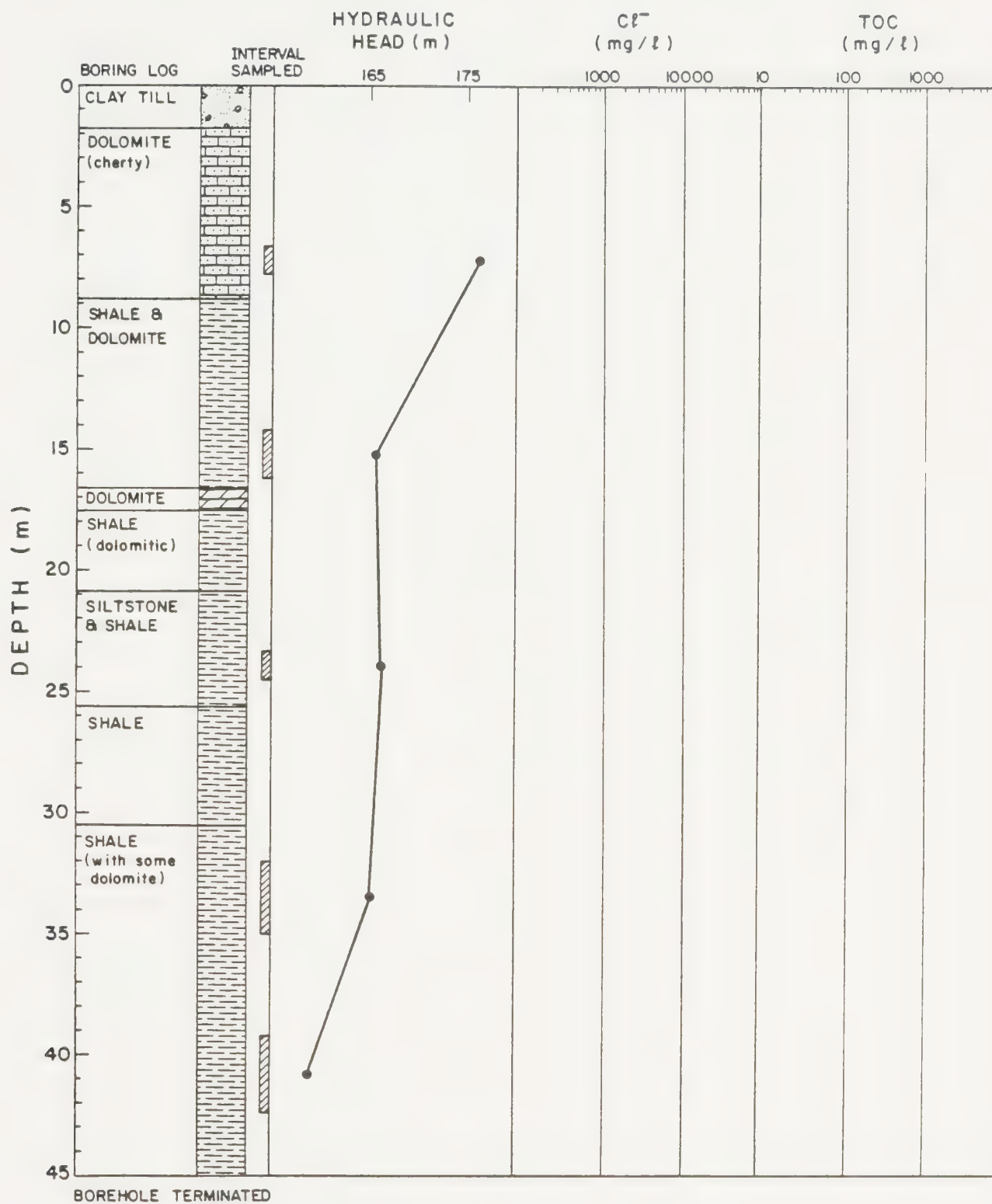


Figure 23. Vertical Profile - UW19

VERTICAL PROFILE - UW 20

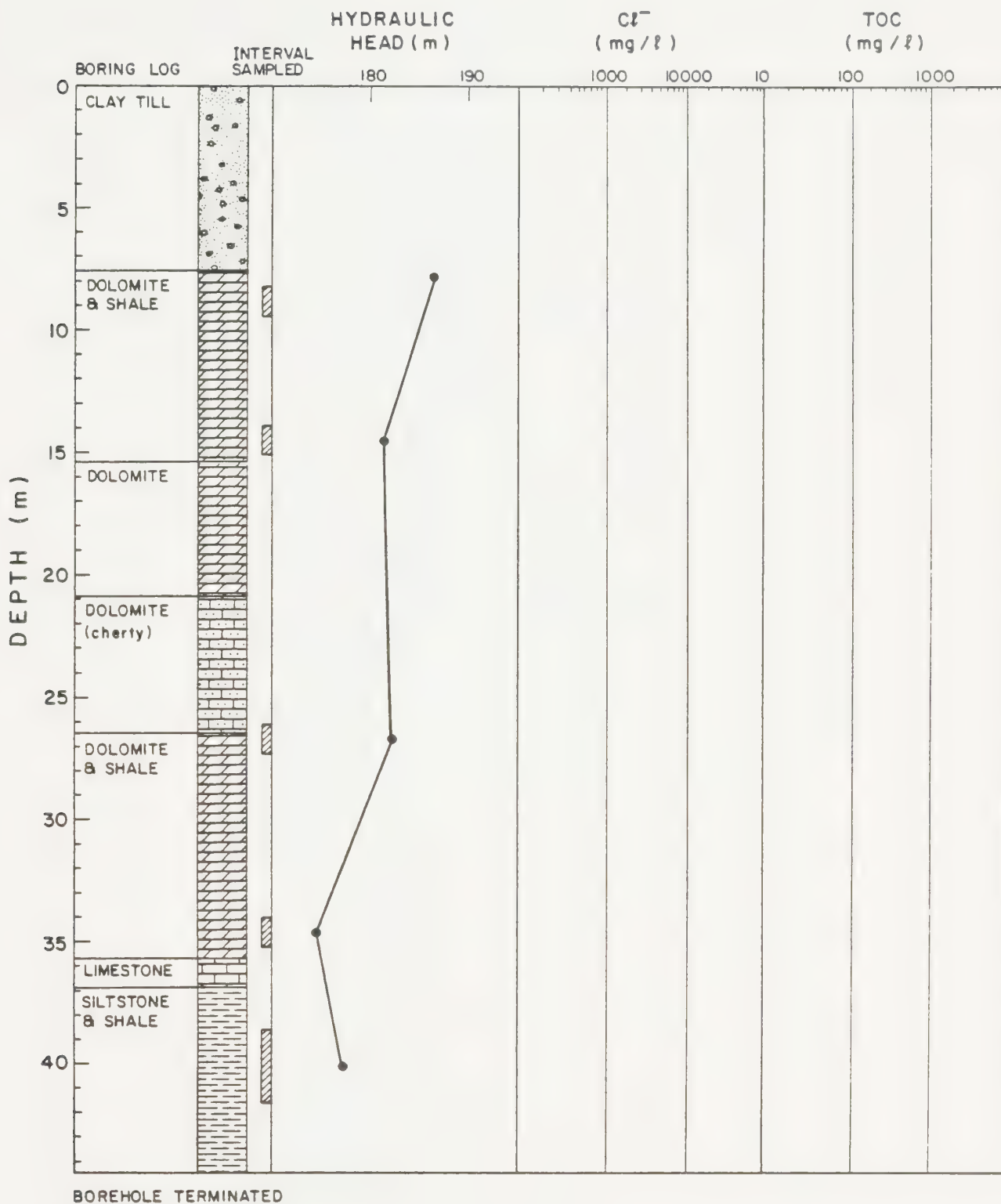


Figure 24. Vertical Profile - UW20

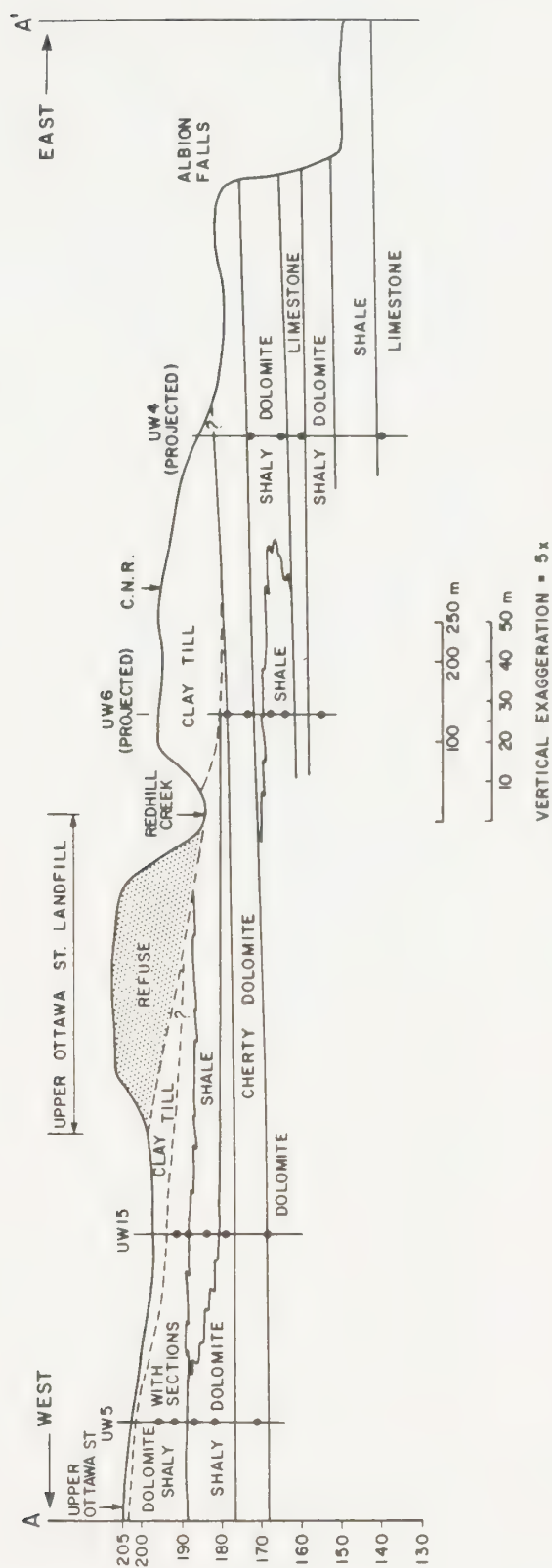


Figure 25. Geologic Cross-Section A-A'

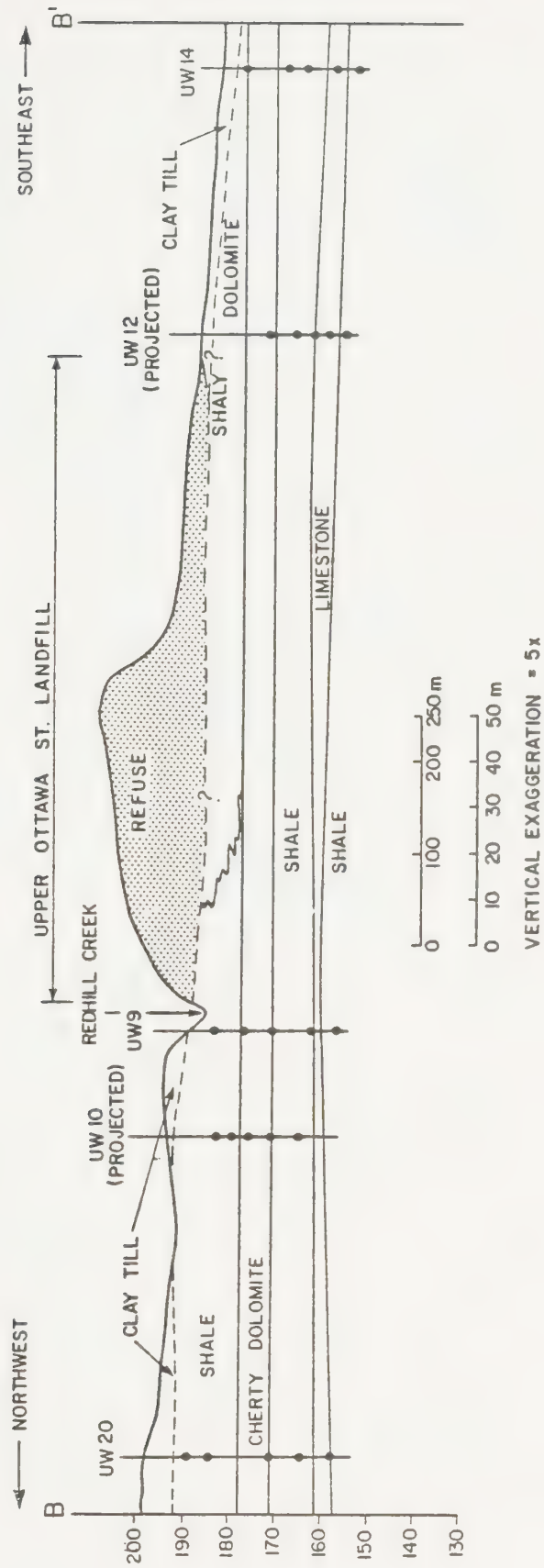


Figure 26. Geologic Cross-Section B-B'



Figure 27. Generalized Water Table Contour Map

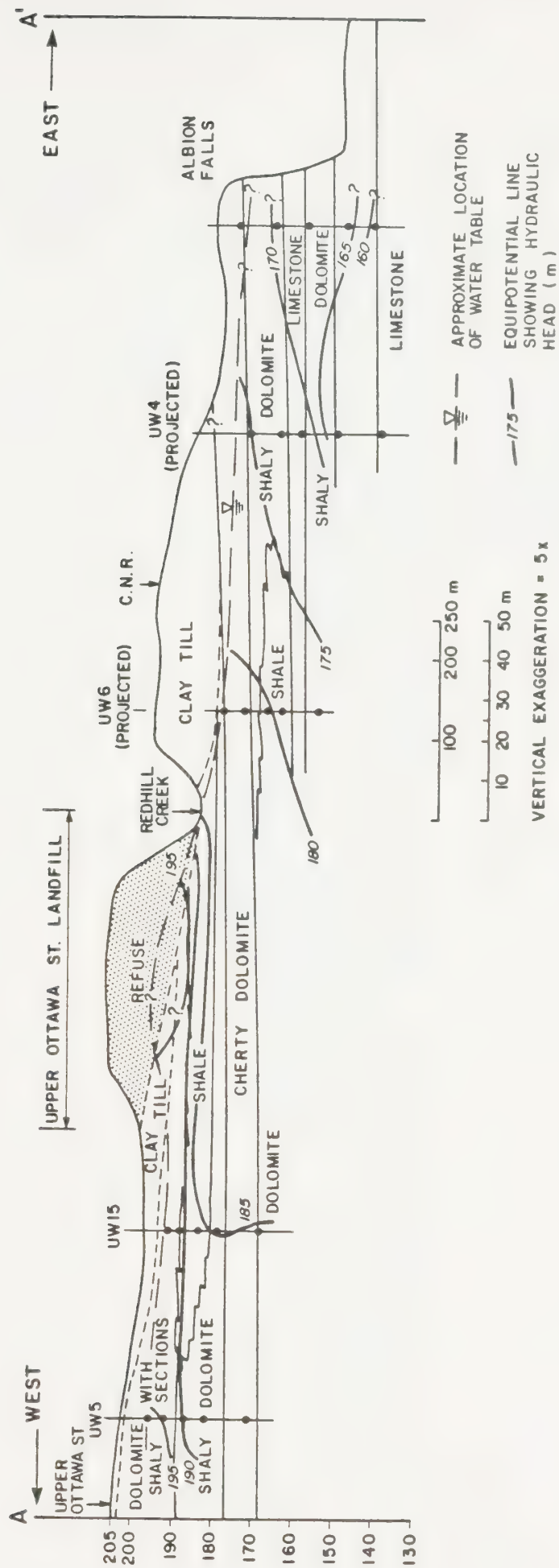


Figure 28. Hydrogeologic Section A-A'

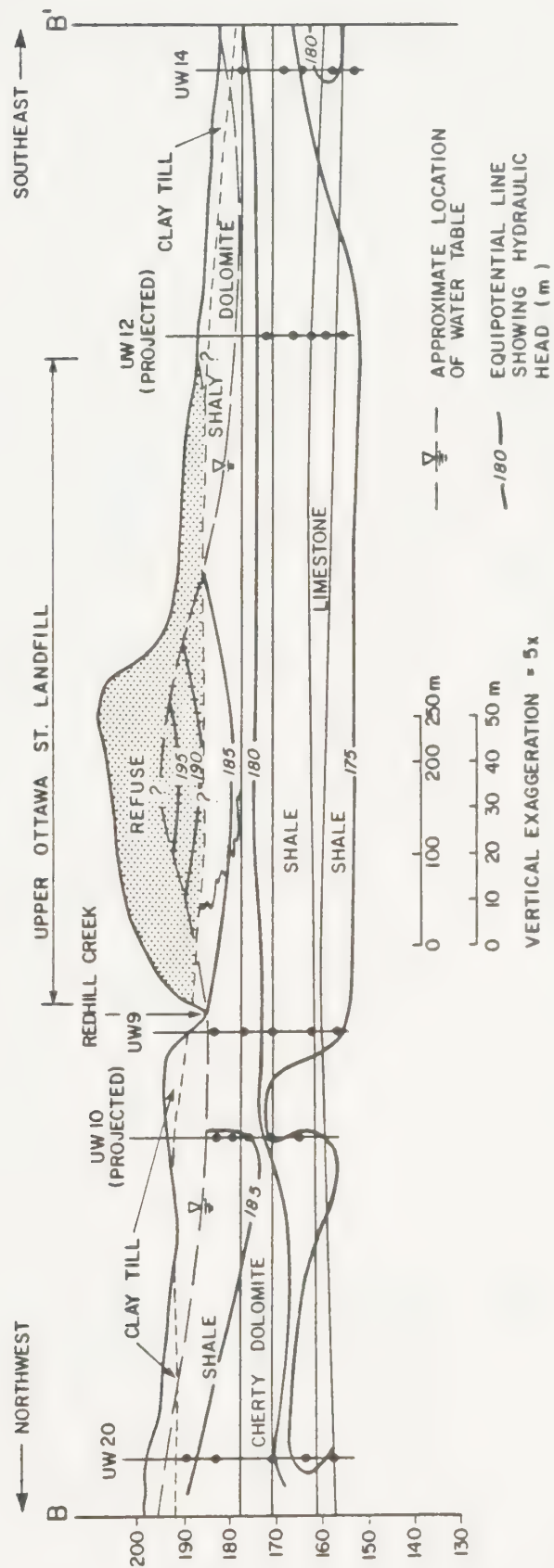


Figure 29. Hydrogeologic Section B-B'

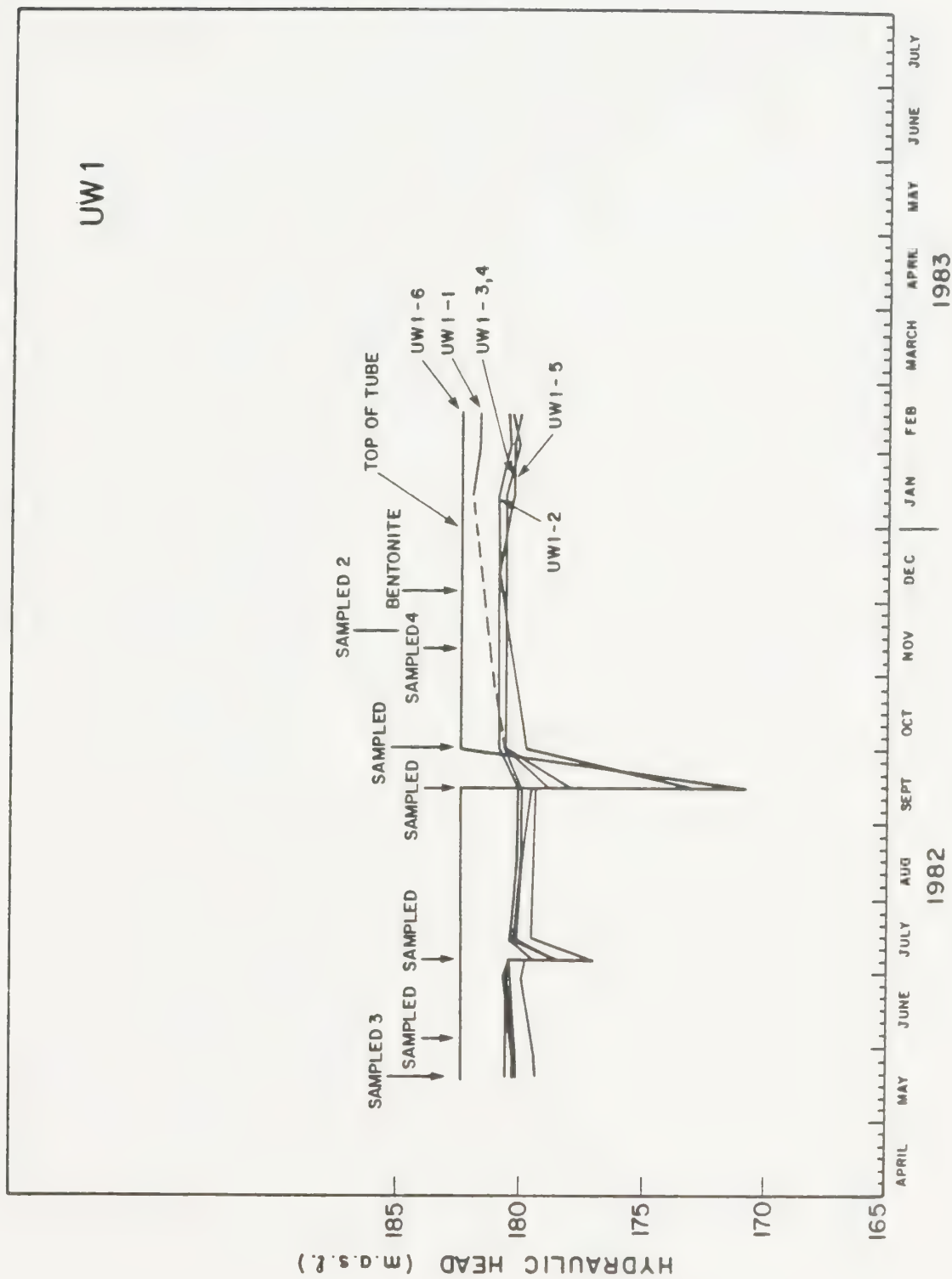


Figure 30. Water-level Hydrographs, UW 1

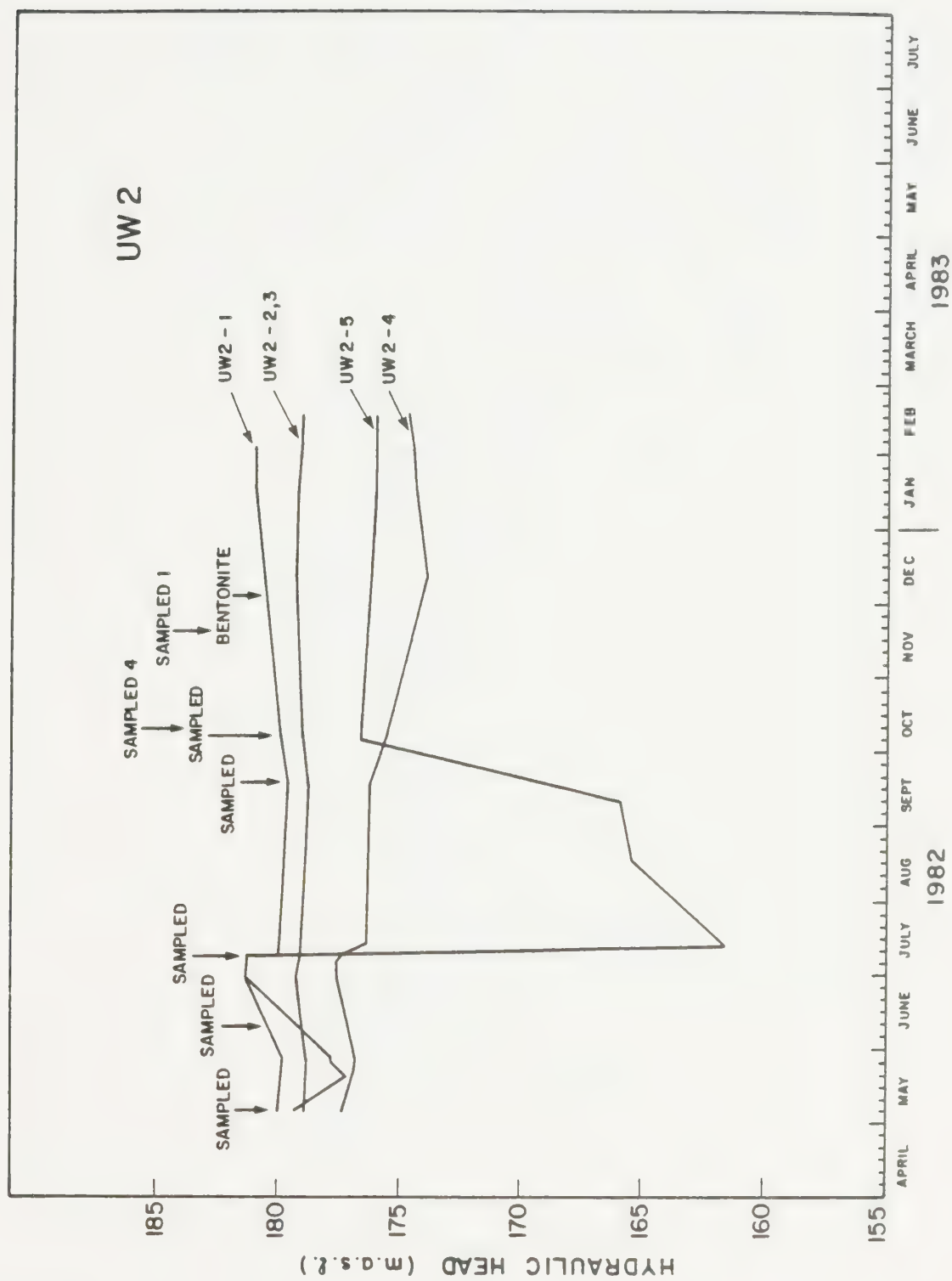


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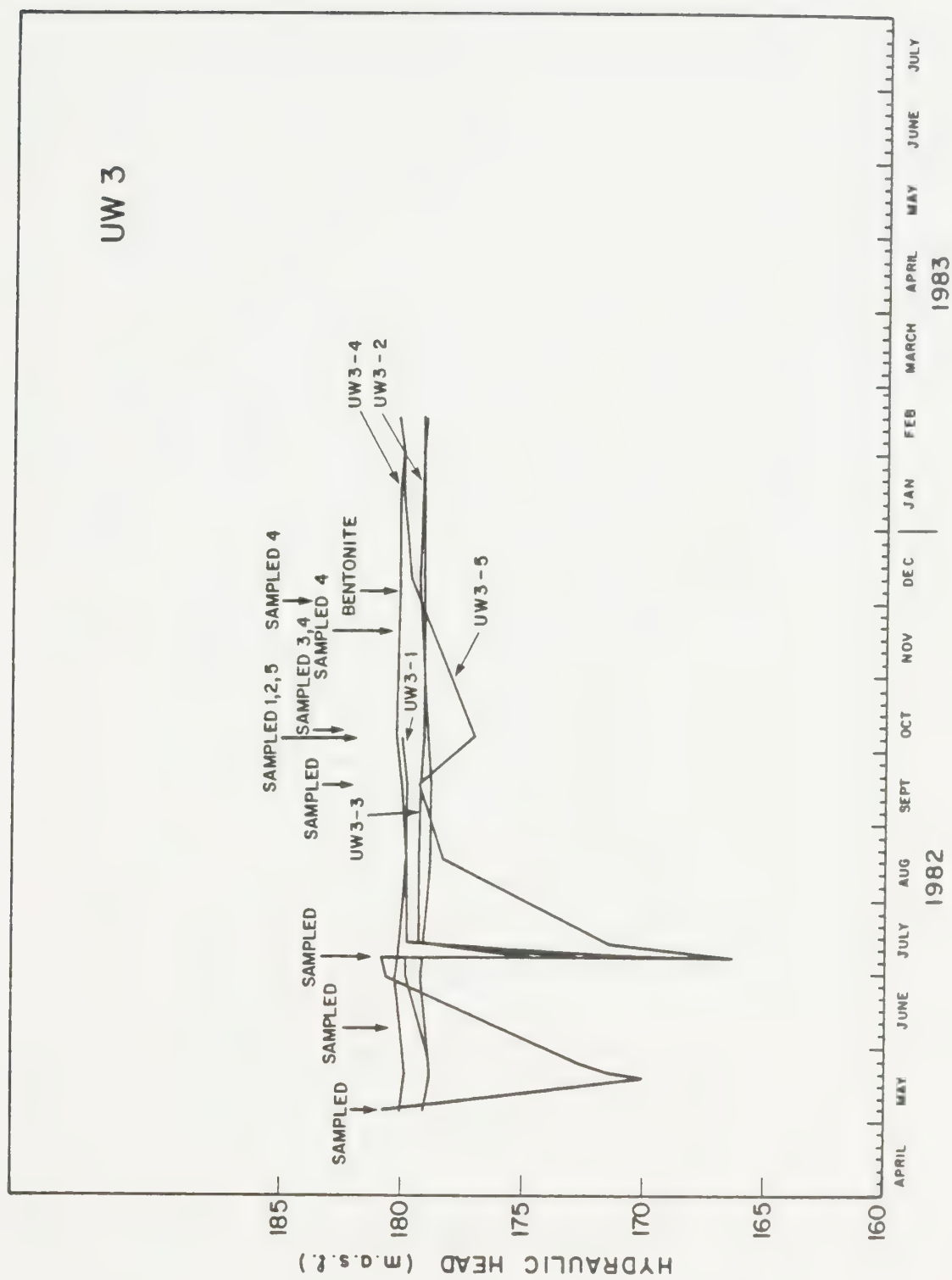


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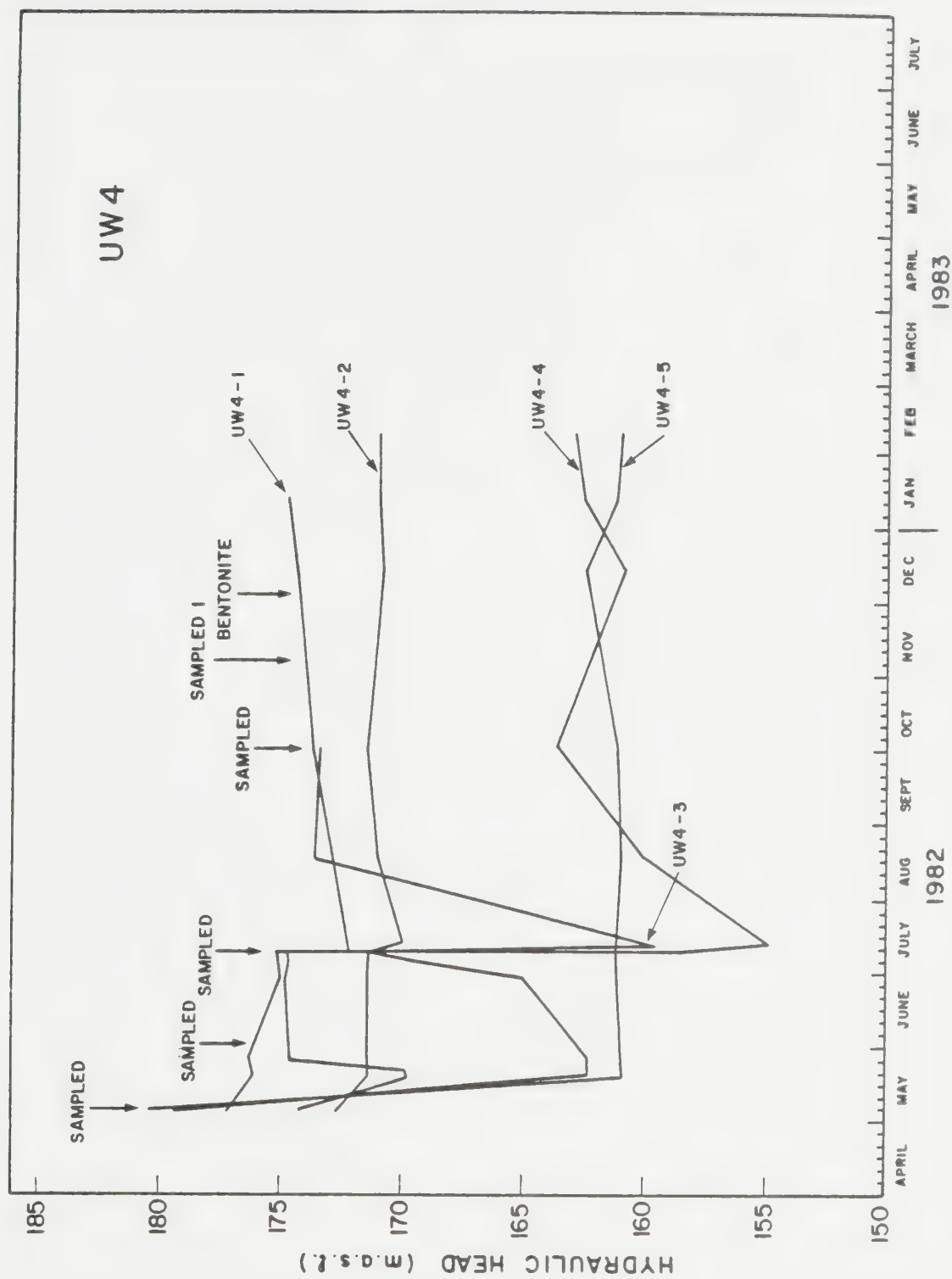


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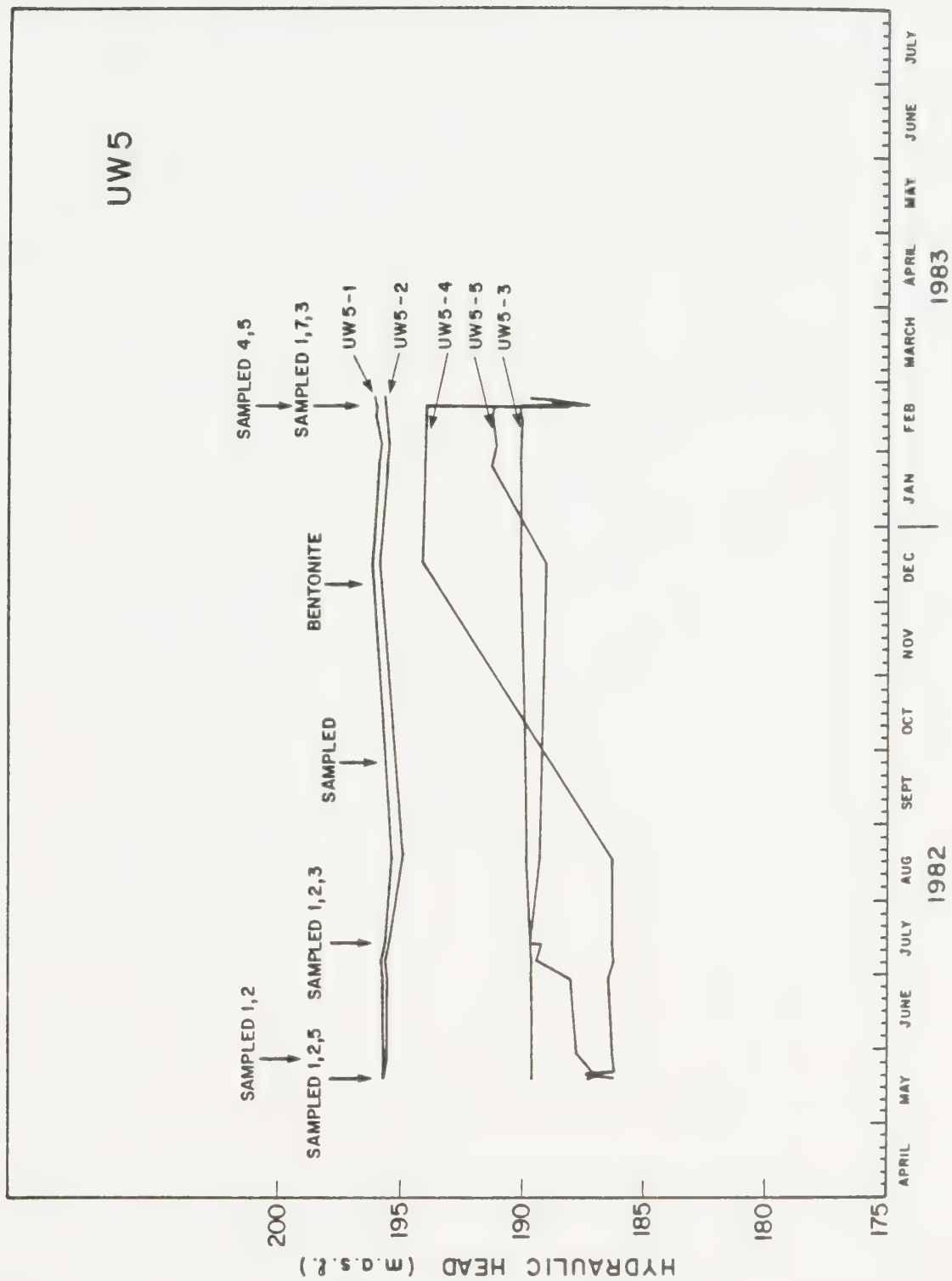


Figure 34. Water-level Hydrographs, UW 5

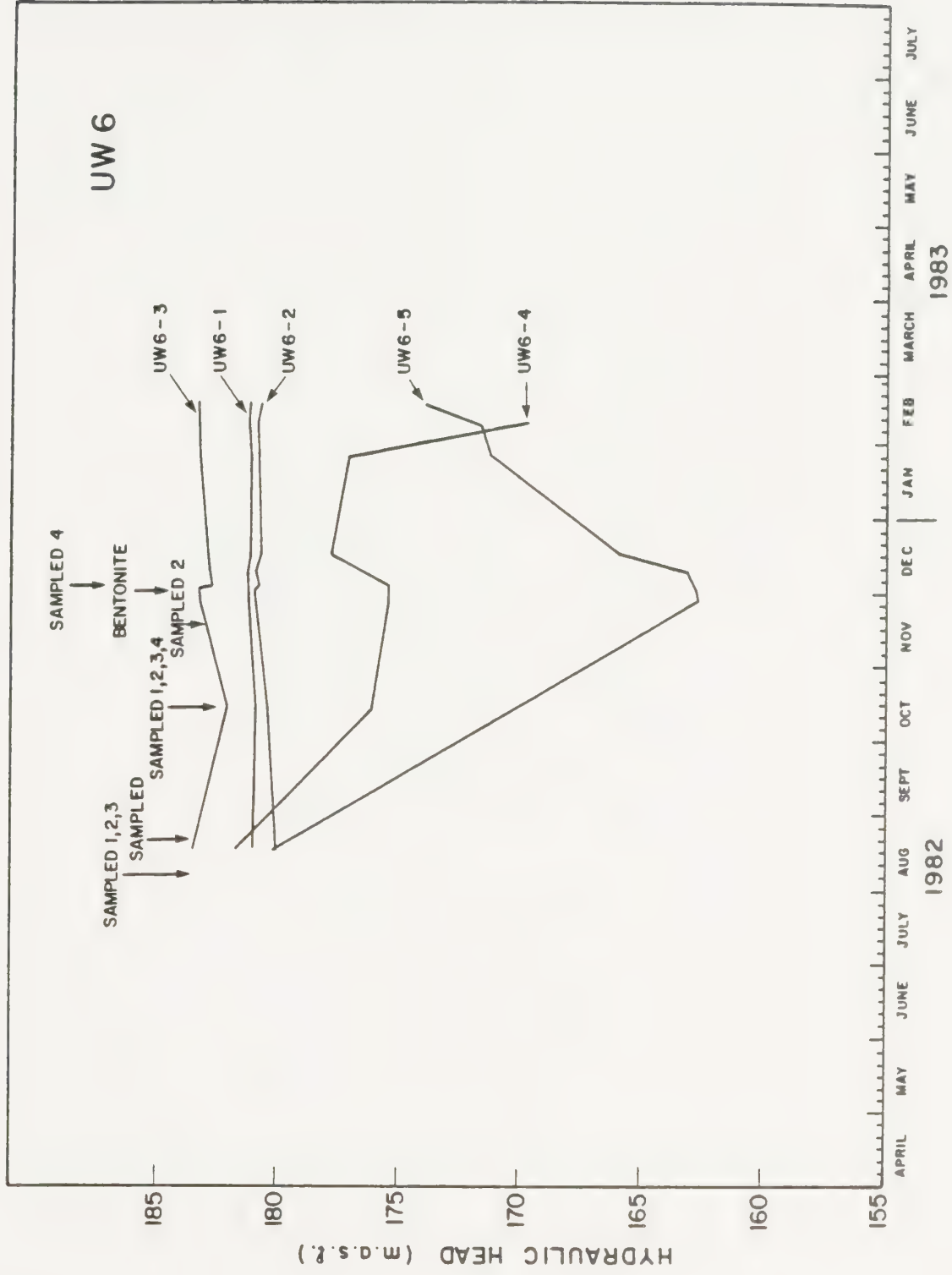


Figure 35. Water-level Hydrographs, UW 6

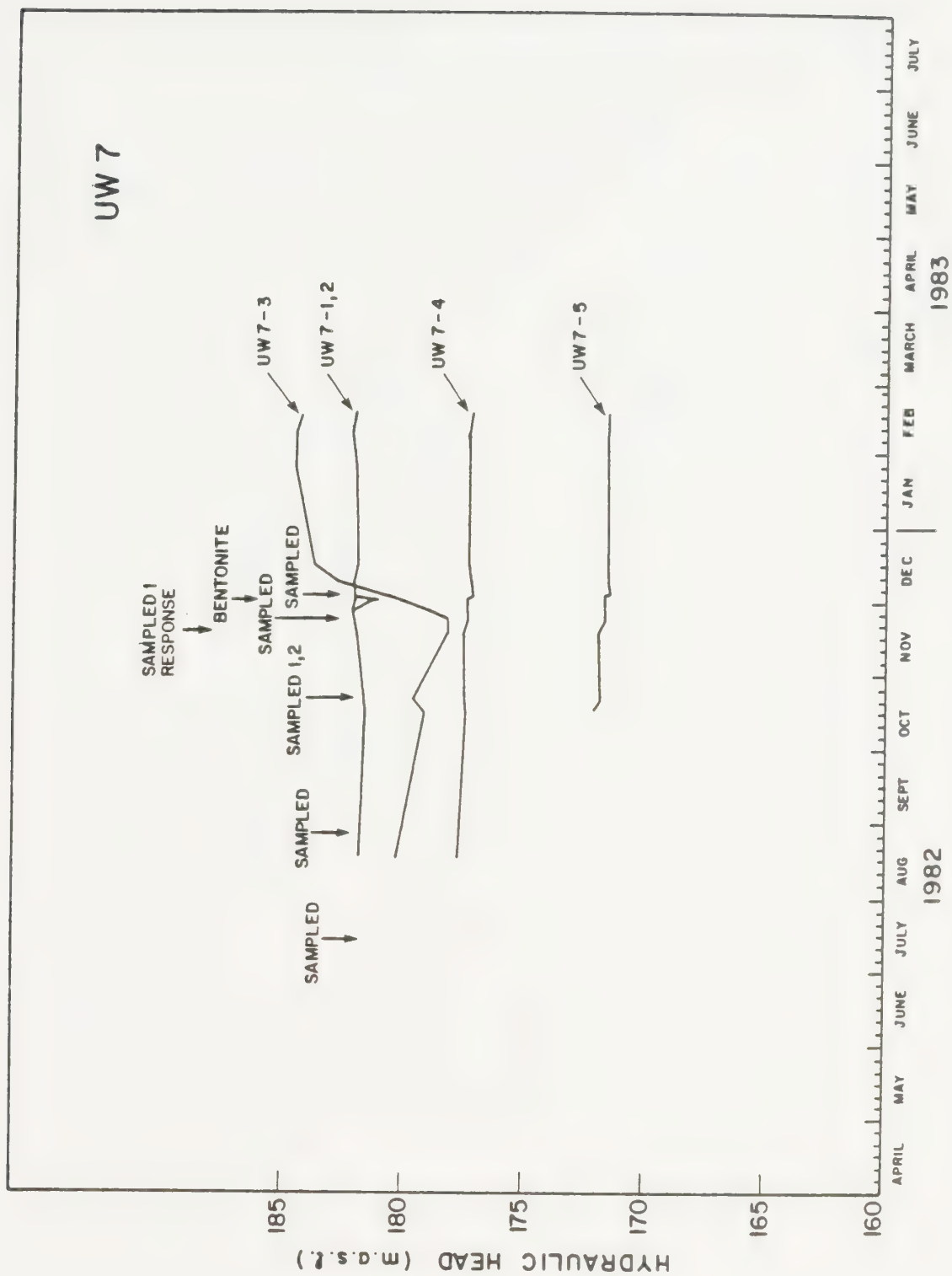


Figure 36. Water-level Hydrographs, UW 7

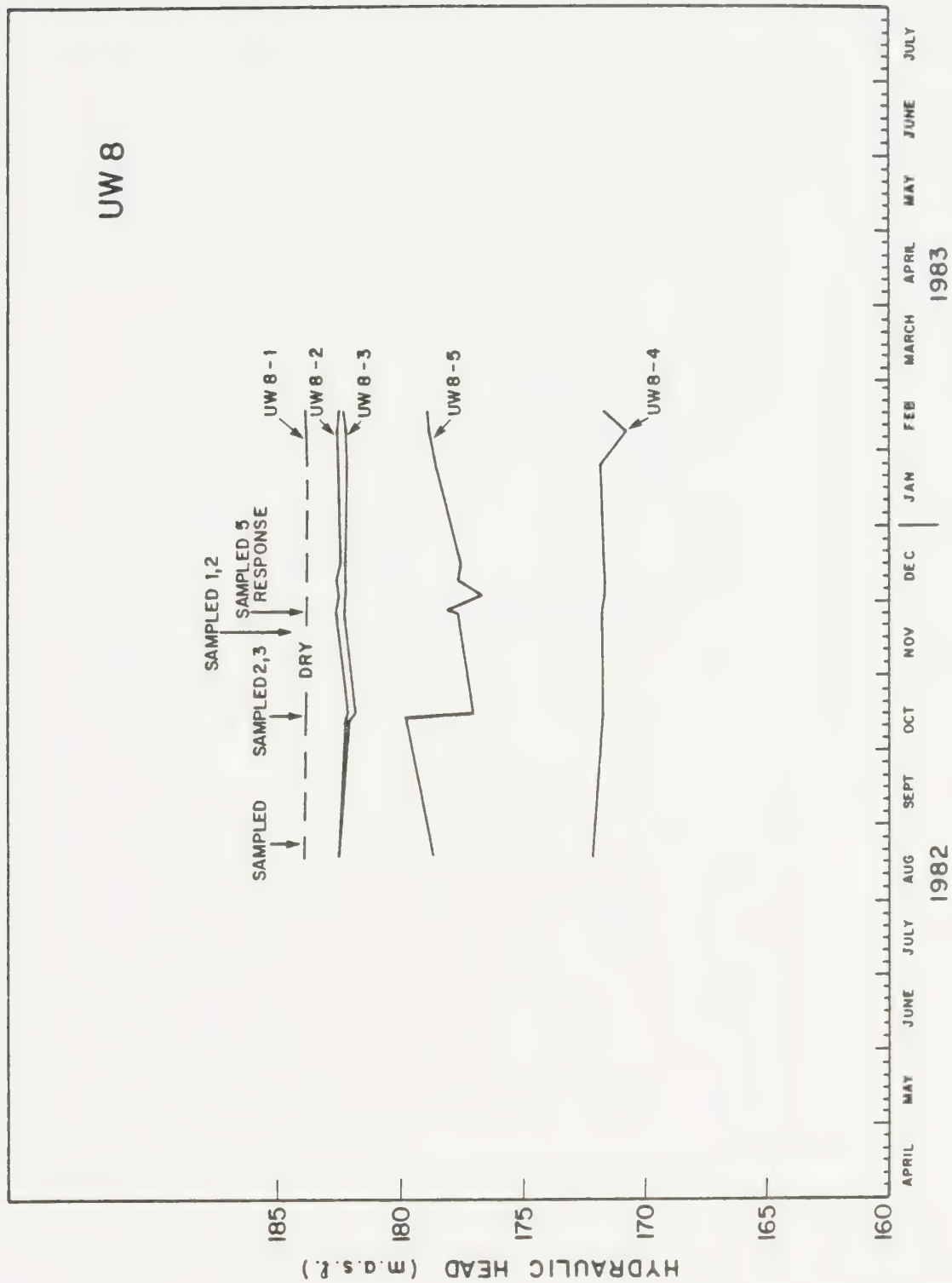


Figure 37. Water-level Hydrographs, UW 8

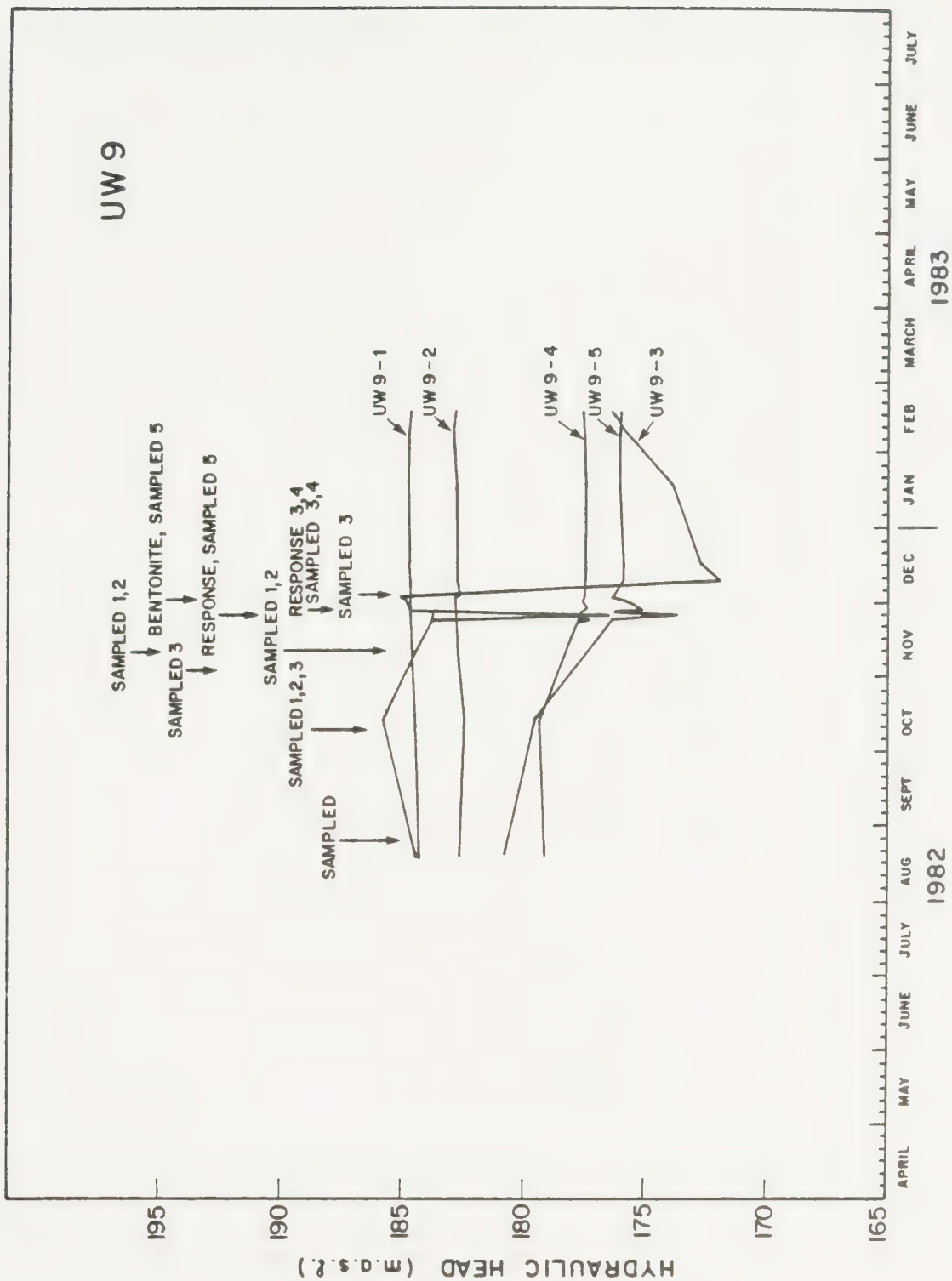


Figure 38. Water-level Hydrographs, UW 9

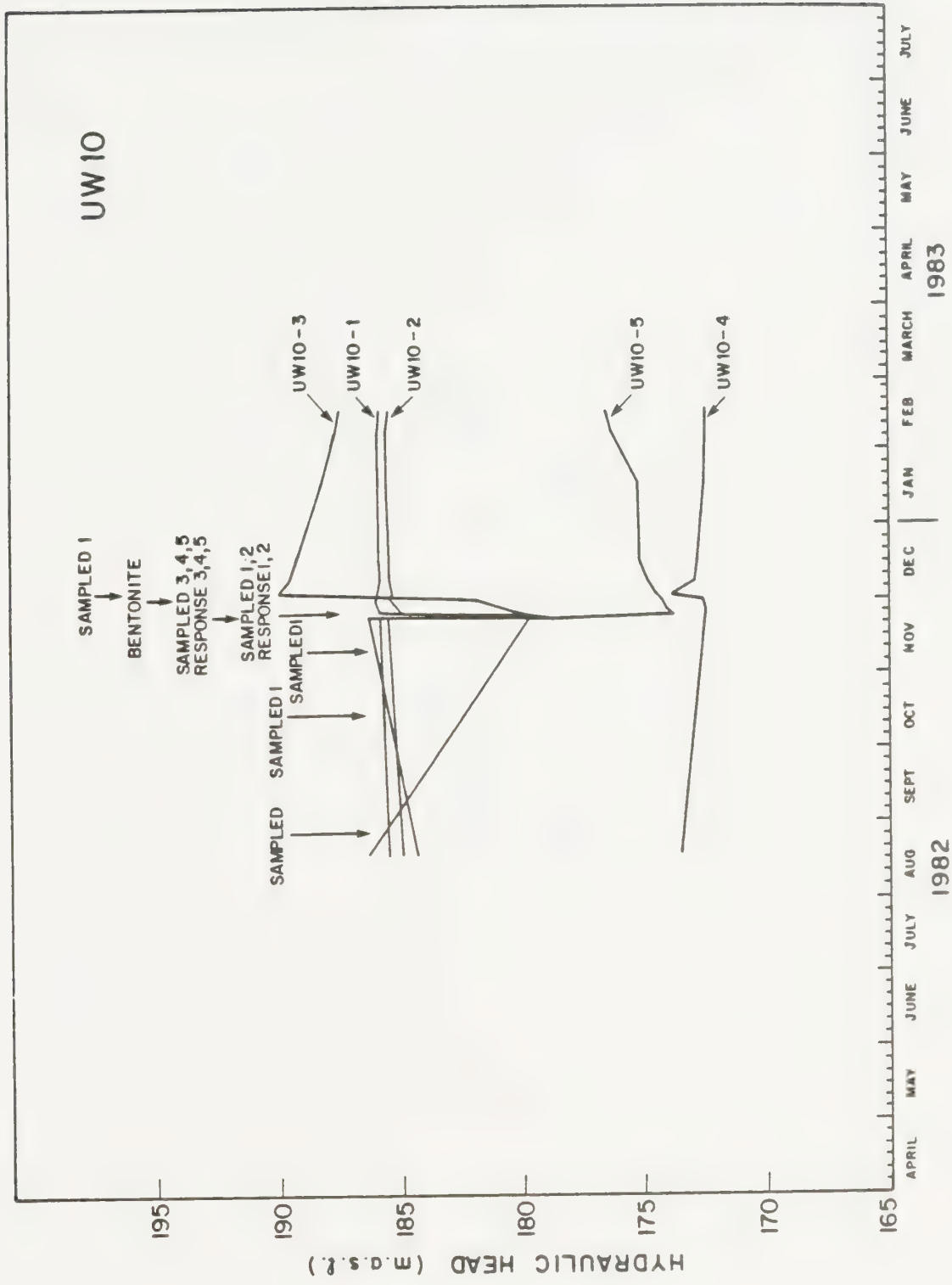


Figure 39. Water-level Hydrographs, UW10

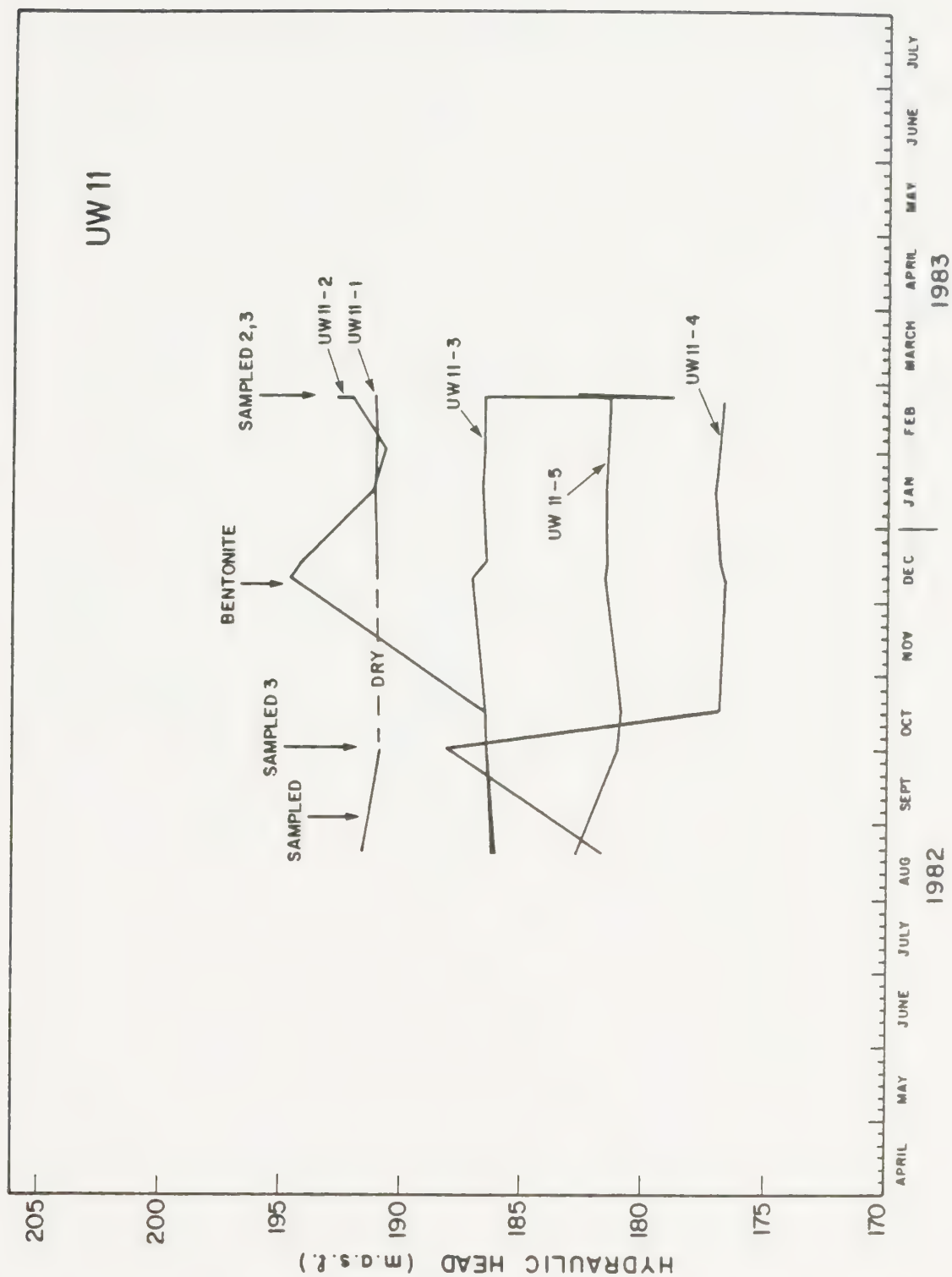


Figure 40. Water-level Hydrographs, UW11

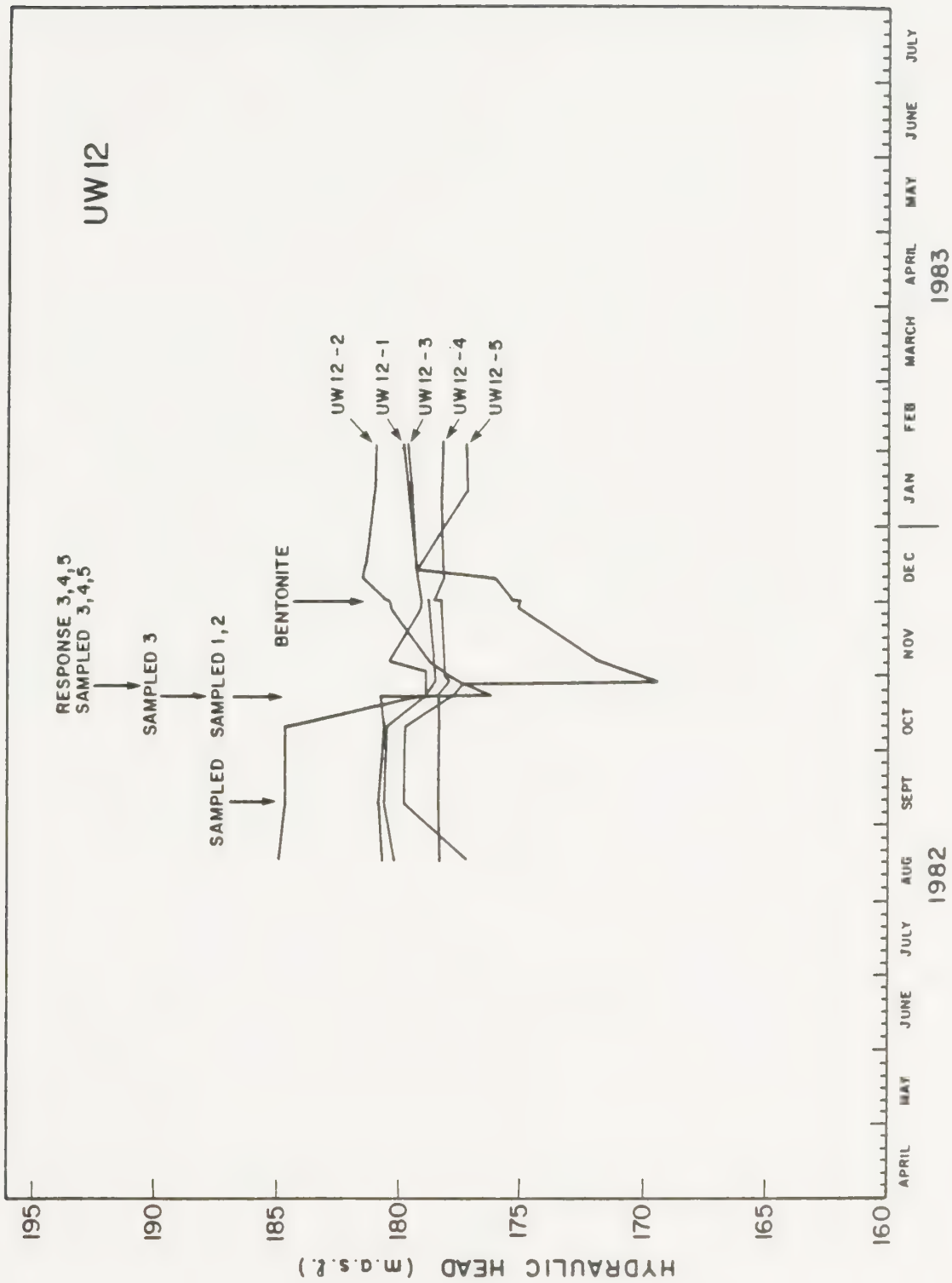


Figure 41. Water-level Hydrographs, UW12

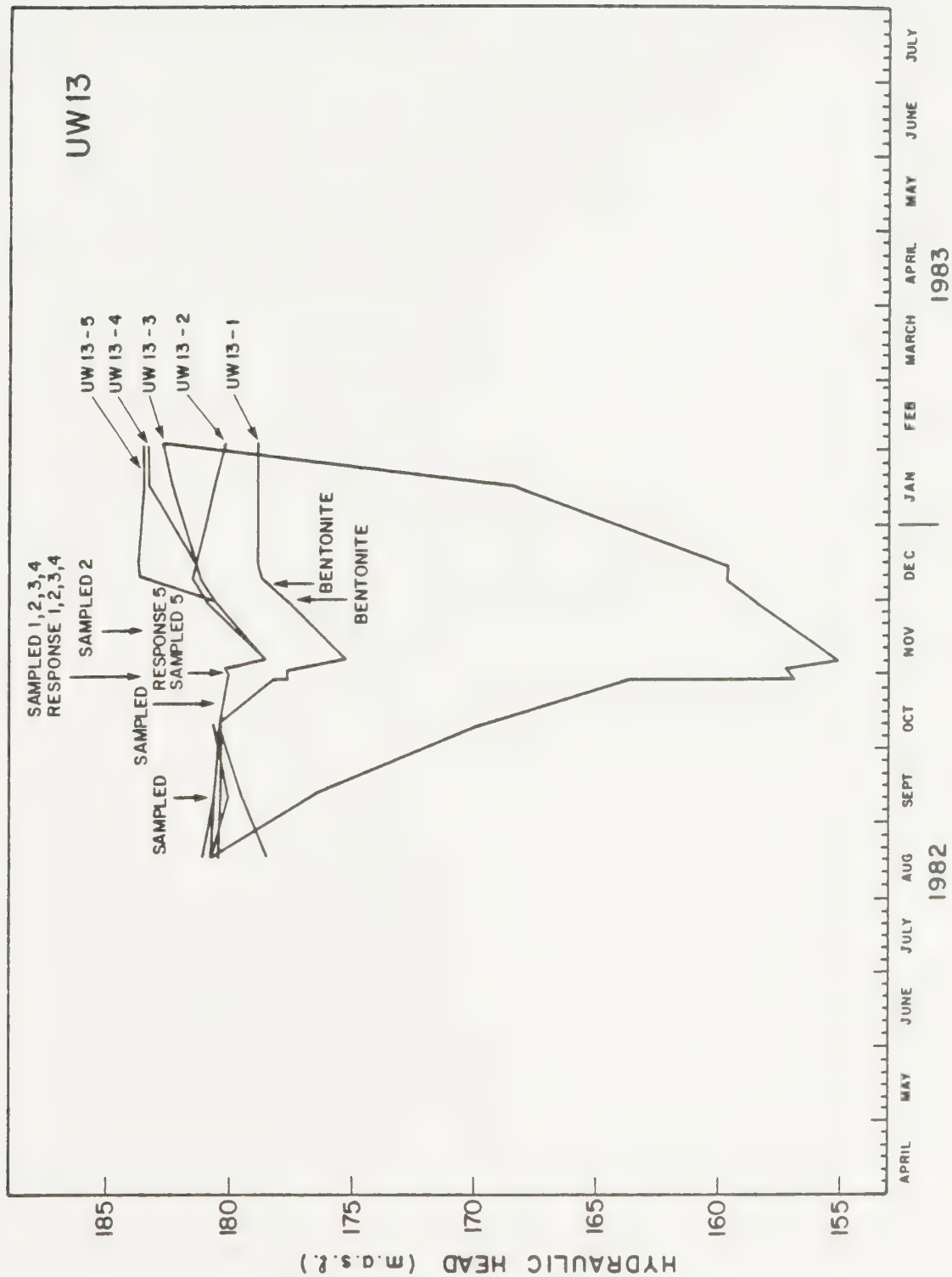


Figure 42. Water-level Hydrographs, UW13

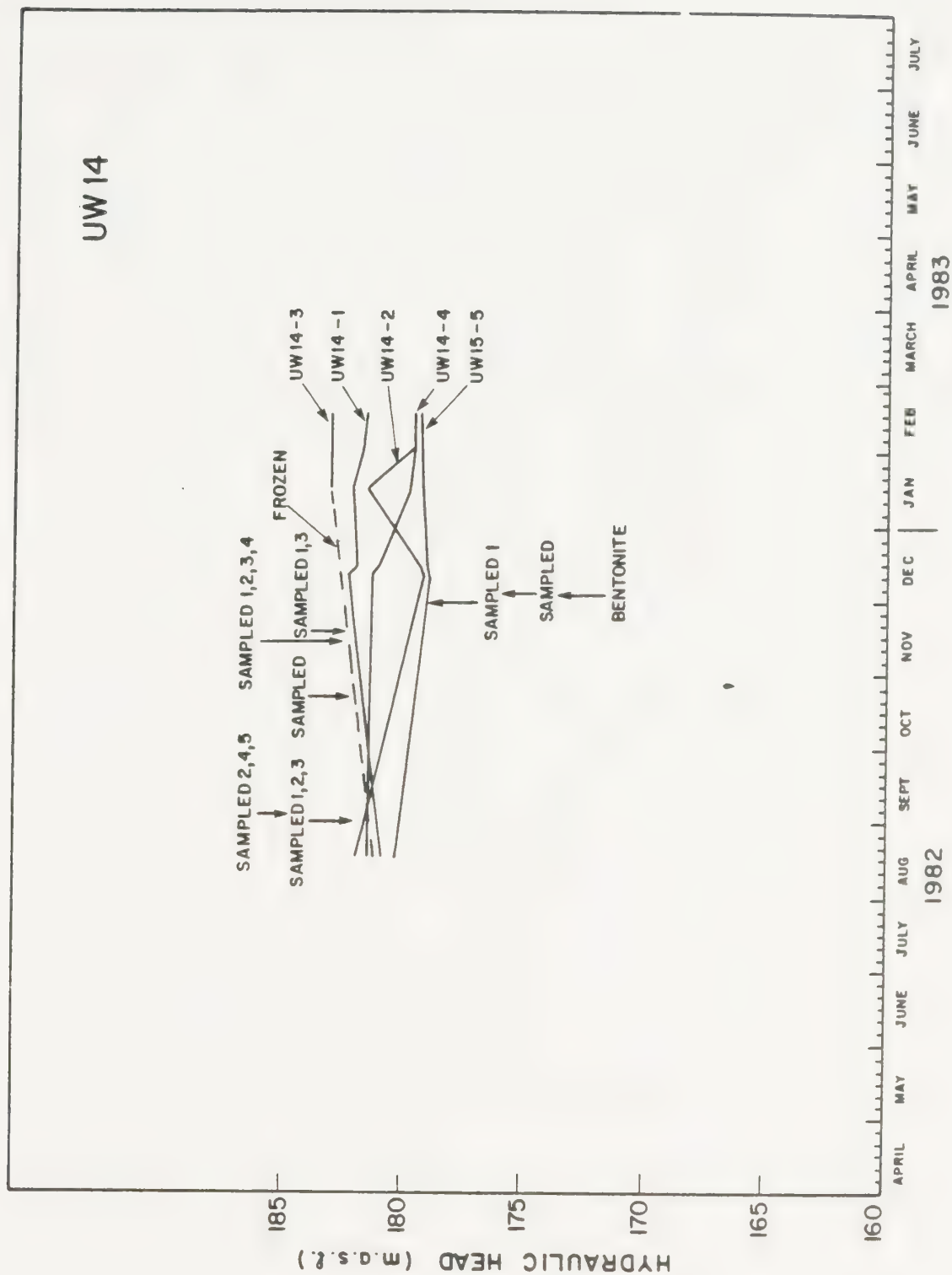


Figure 43. Water-level Hydrographs, UW14

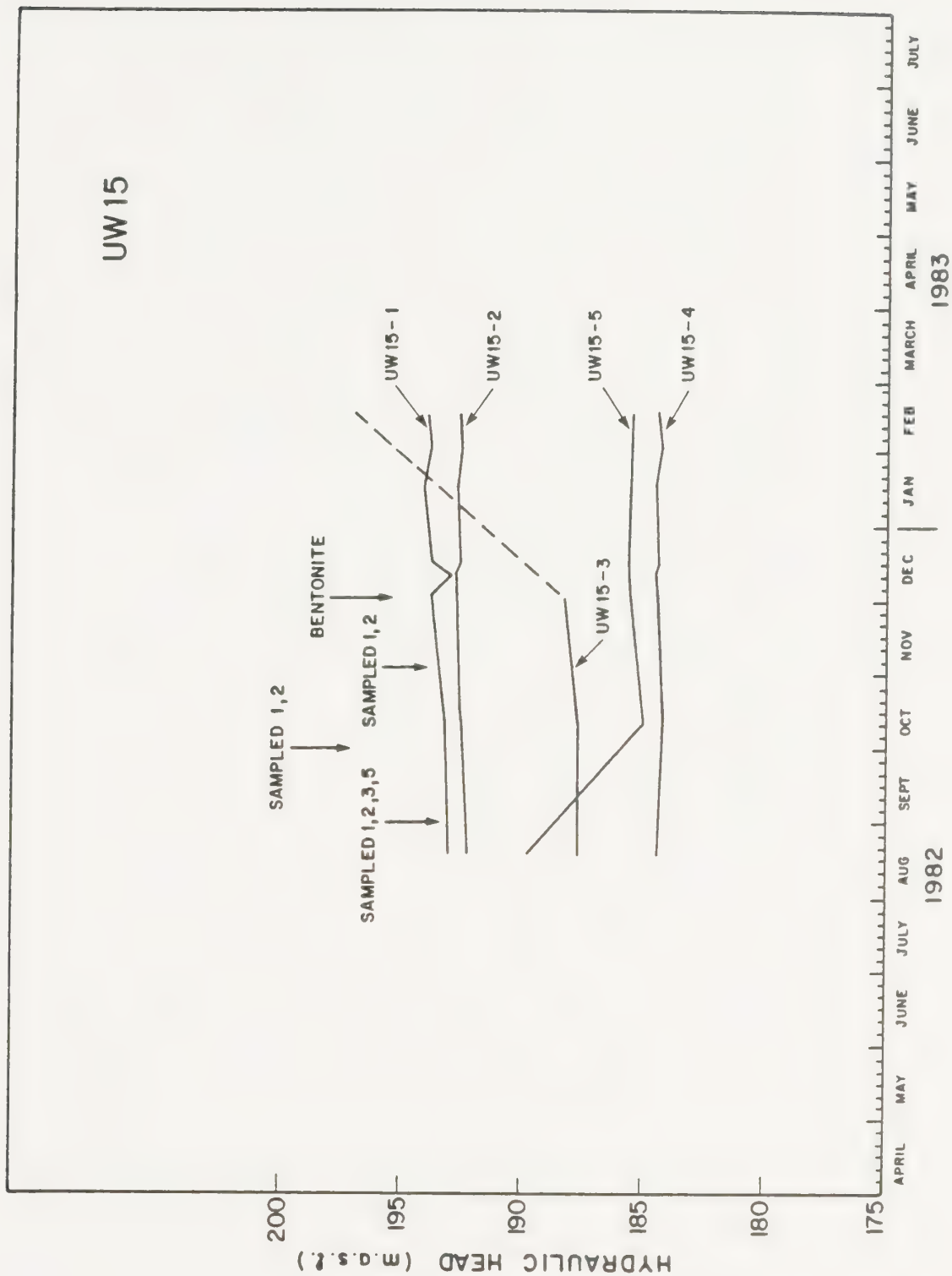


Figure 44. Water-level Hydrographs, UW15

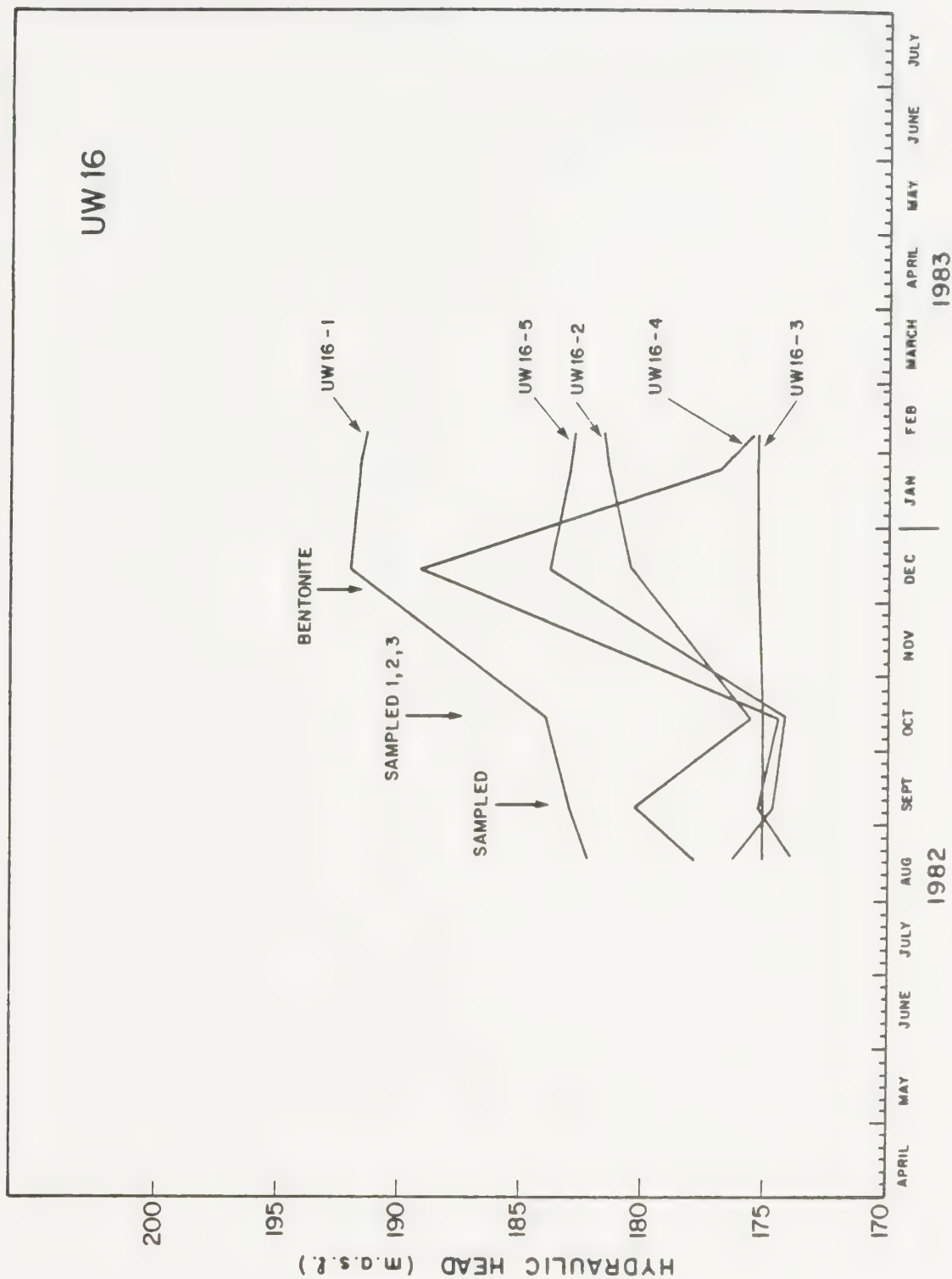


Figure 45. Water-level Hydrographs, UW16

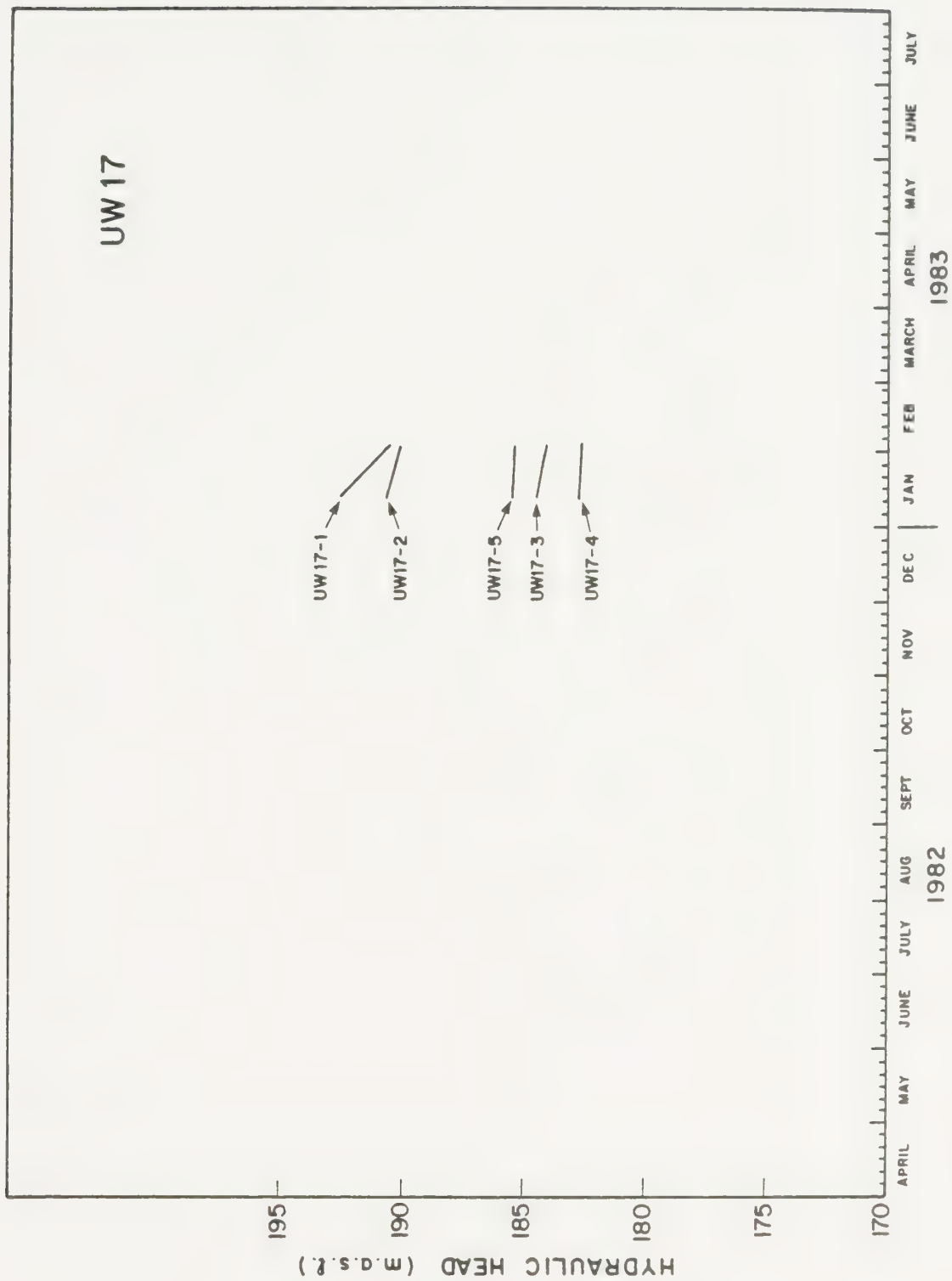


Figure 46. Water-level Hydrographs, UW17

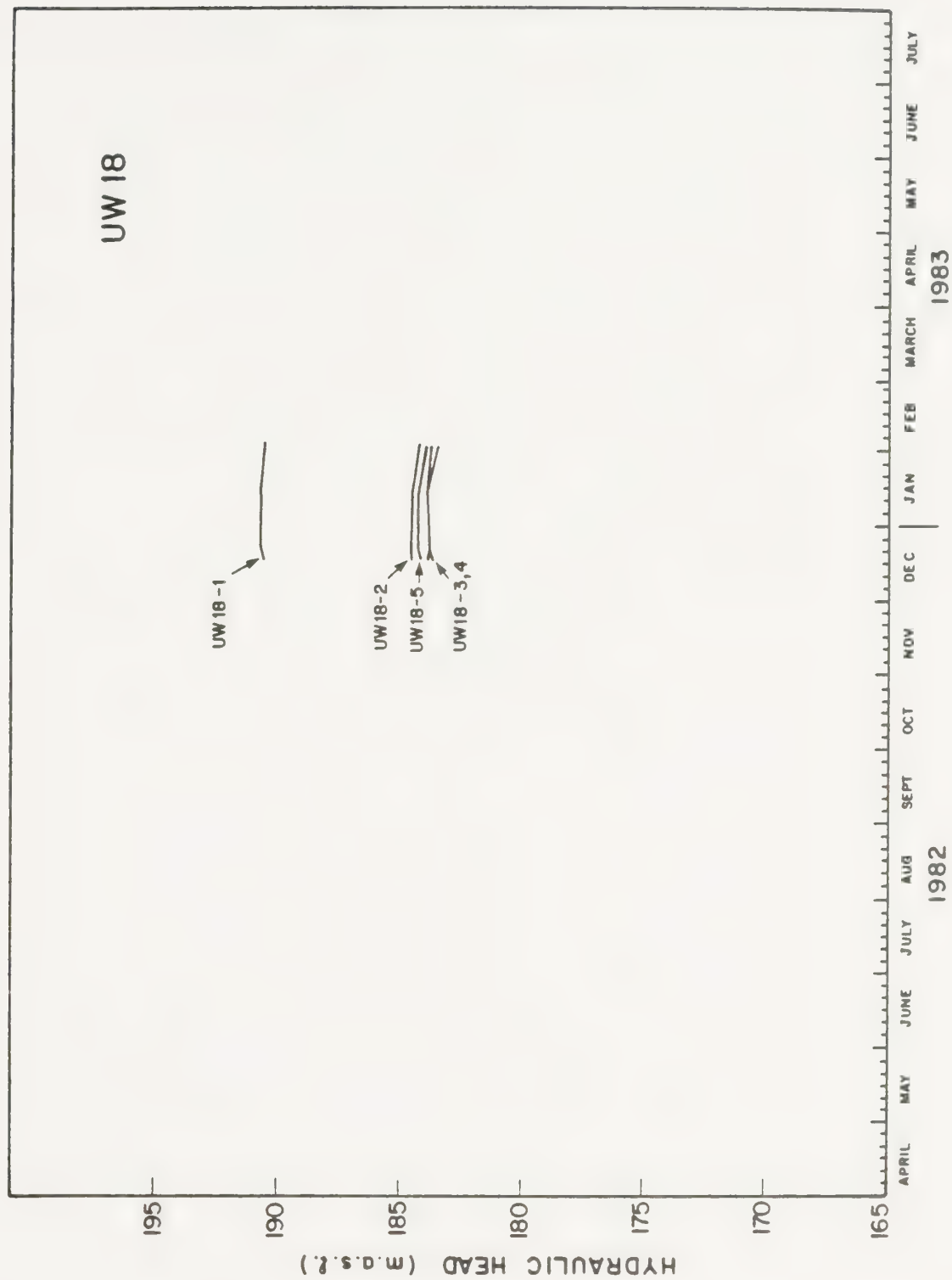


Figure 47. Water-level Hydrographs, UW18

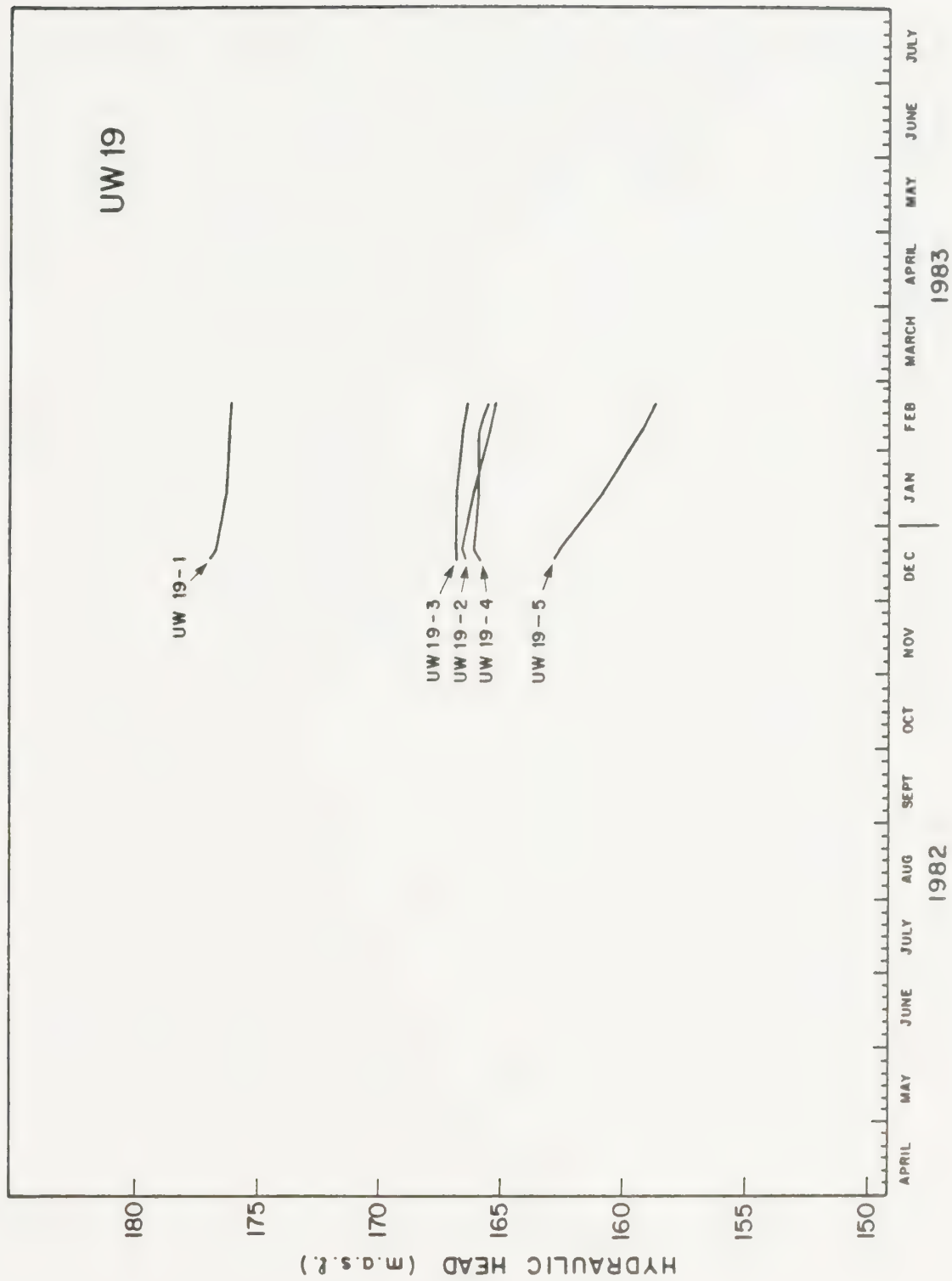


Figure 48. Water-level Hydrographs, UW19

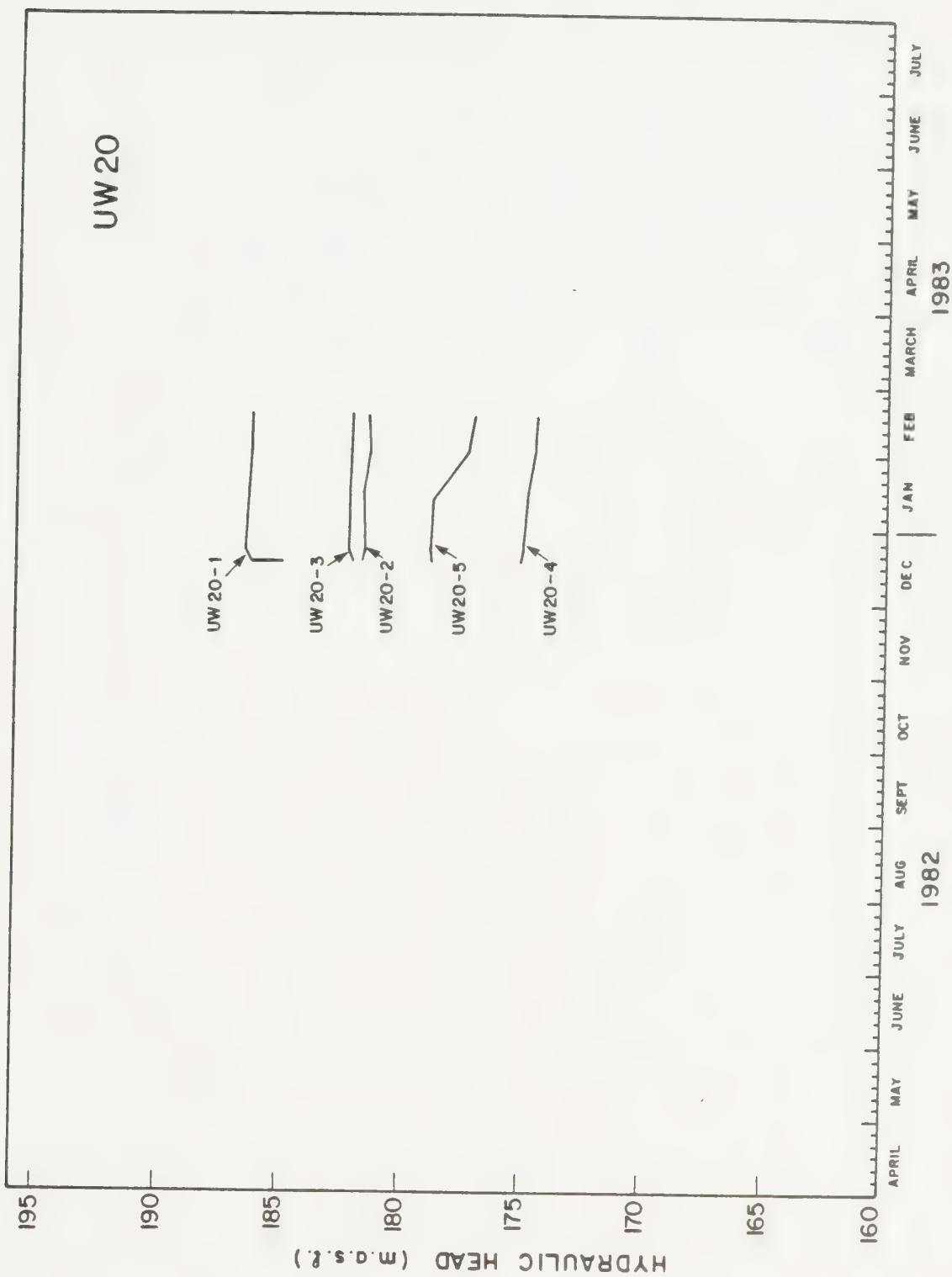


Figure 49. Water-level Hydrographs, UW20

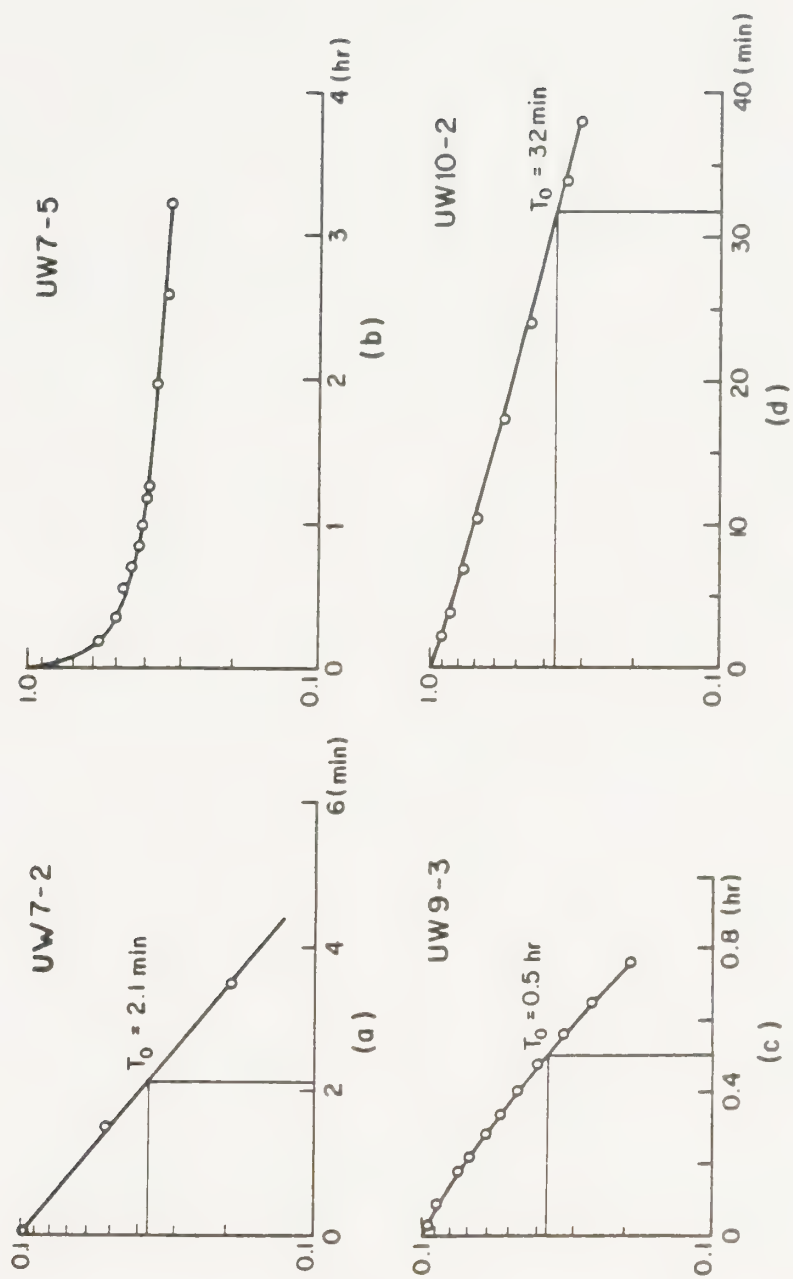


Figure 50. Groundwater response curves

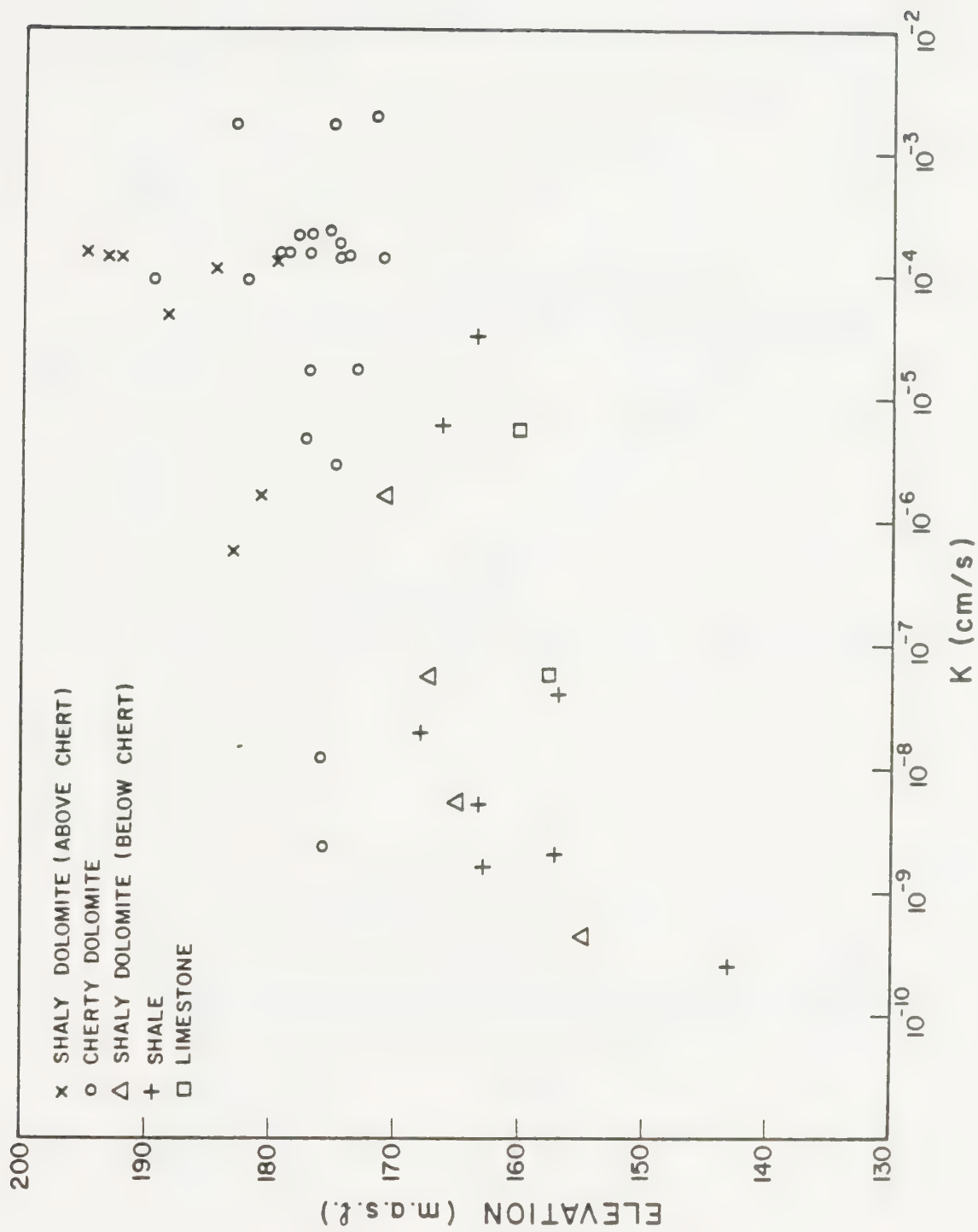


Figure 51. Hydraulic conductivity data

VERTICAL PROFILE - UW 2

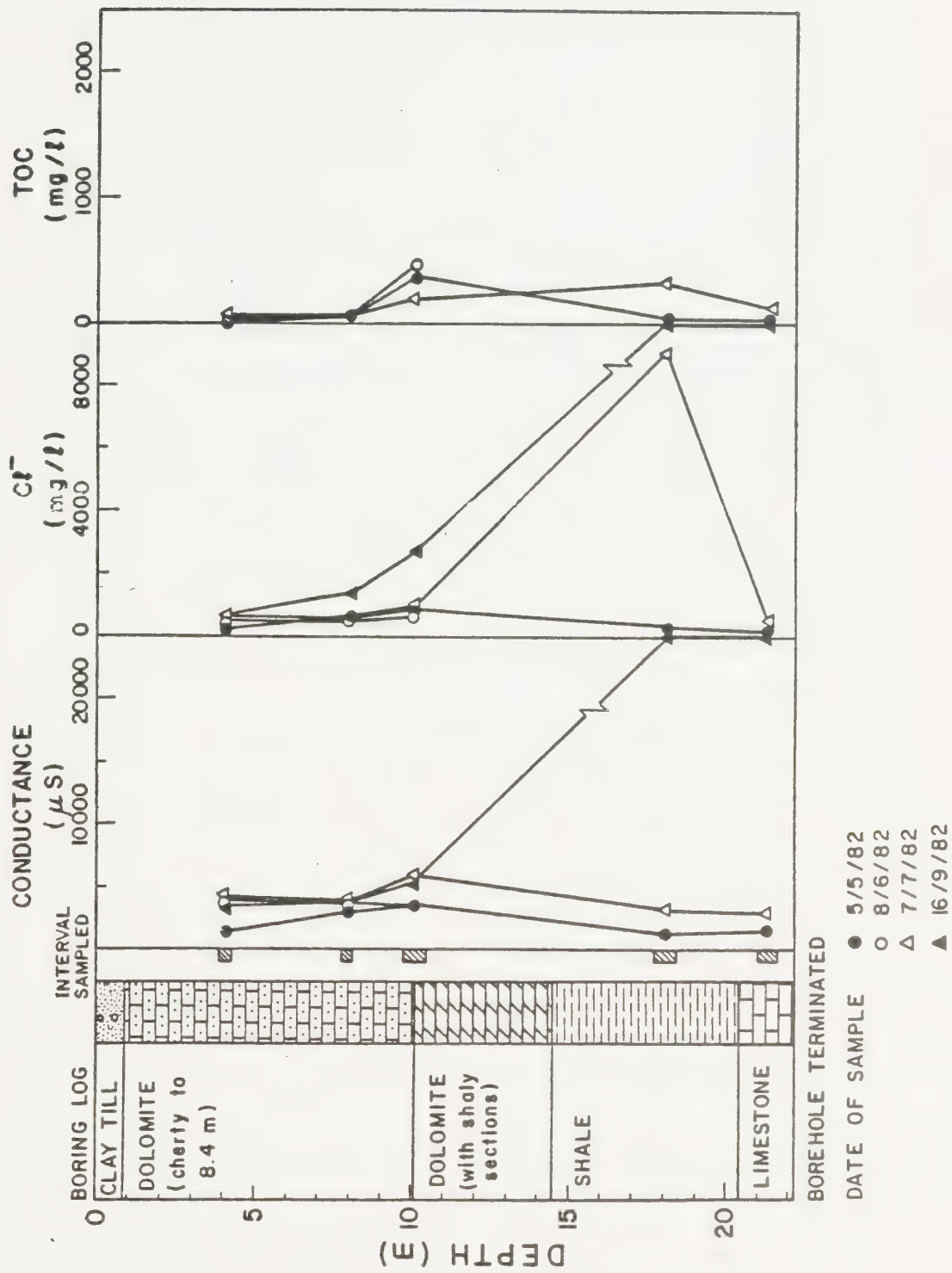


Figure 52. Stabilization of routine chemical parameters at UW 2

Figure 53. Variation of selected geochemical parameters with well volumes removed - UW 9-1 and UW10-1.

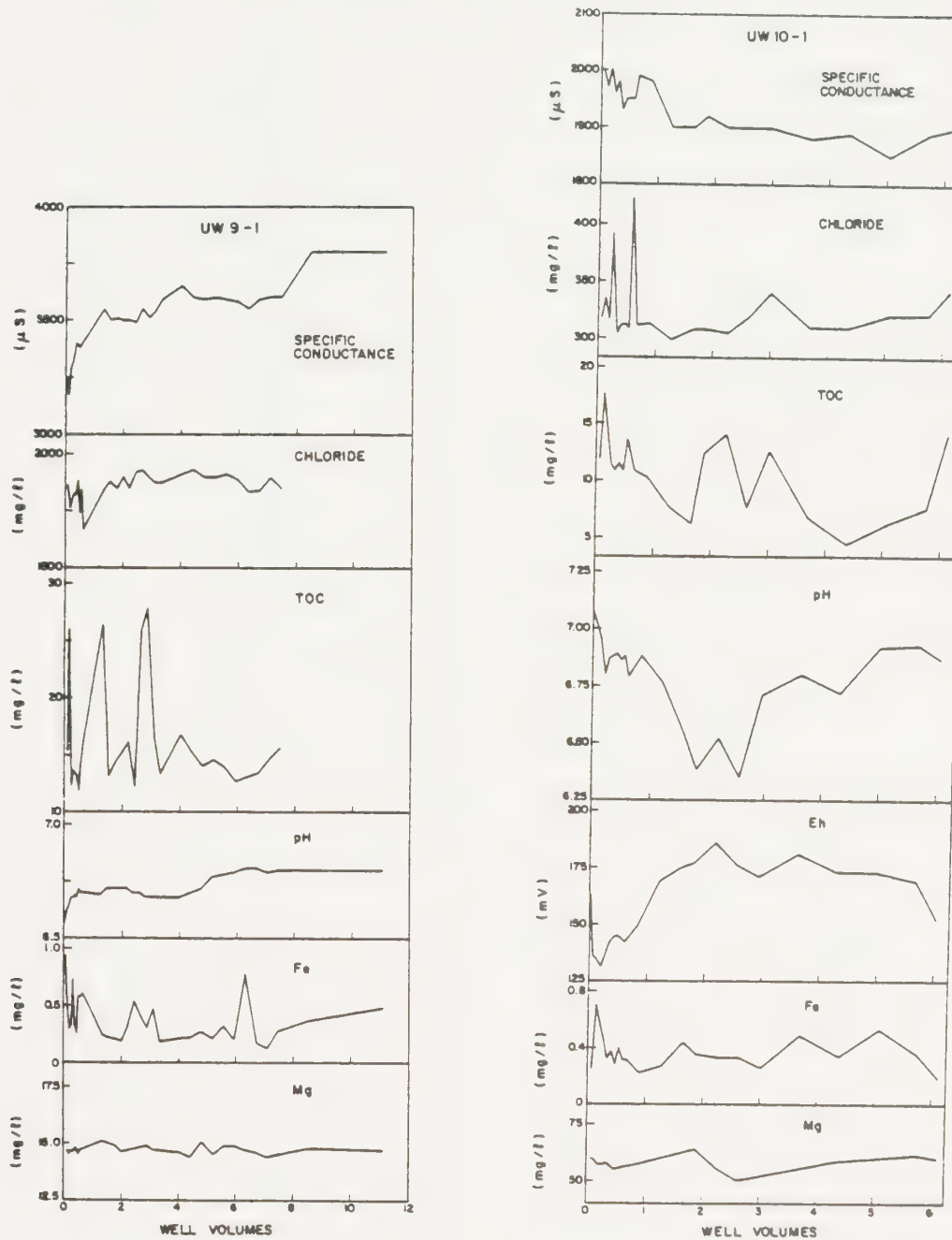


Figure 54. Representative gas chromatograms of volatile, halogenated organic compounds.

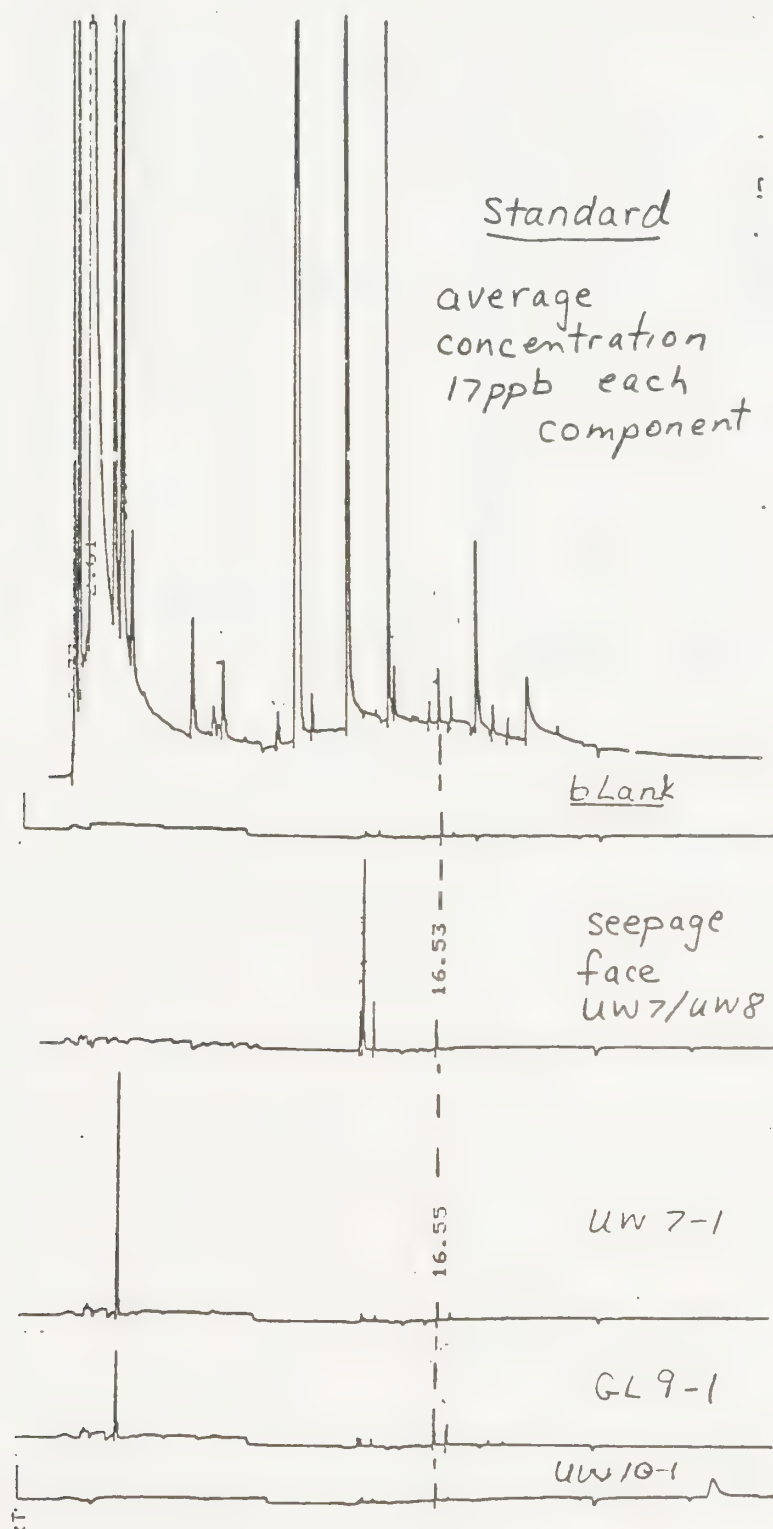


Figure 55. Representative gas chromatograms of volatile organic compounds. Note that the peak heights of the standard have been attenuated 8x relative to the blank and samples.

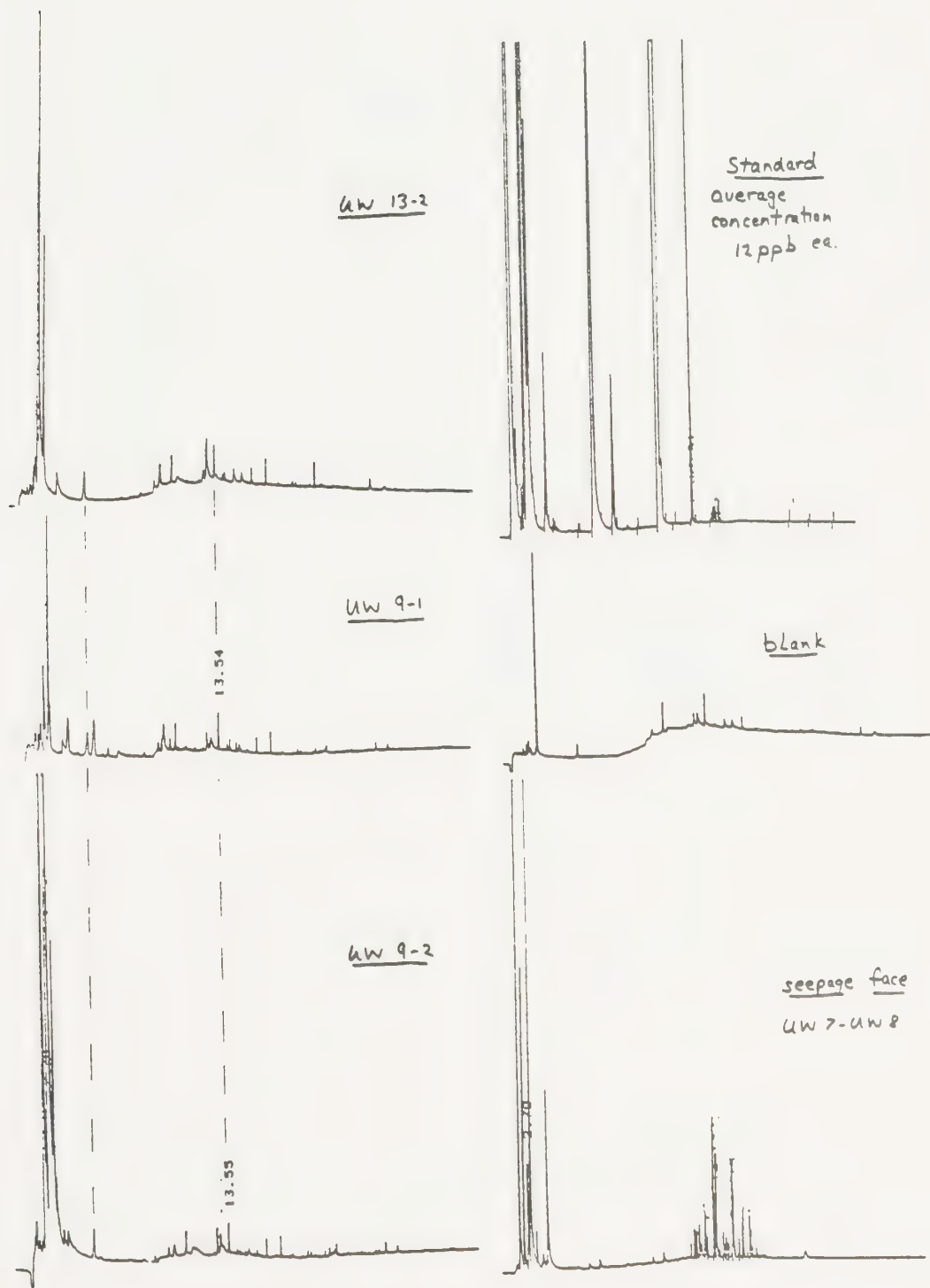


Figure 55. Continued.

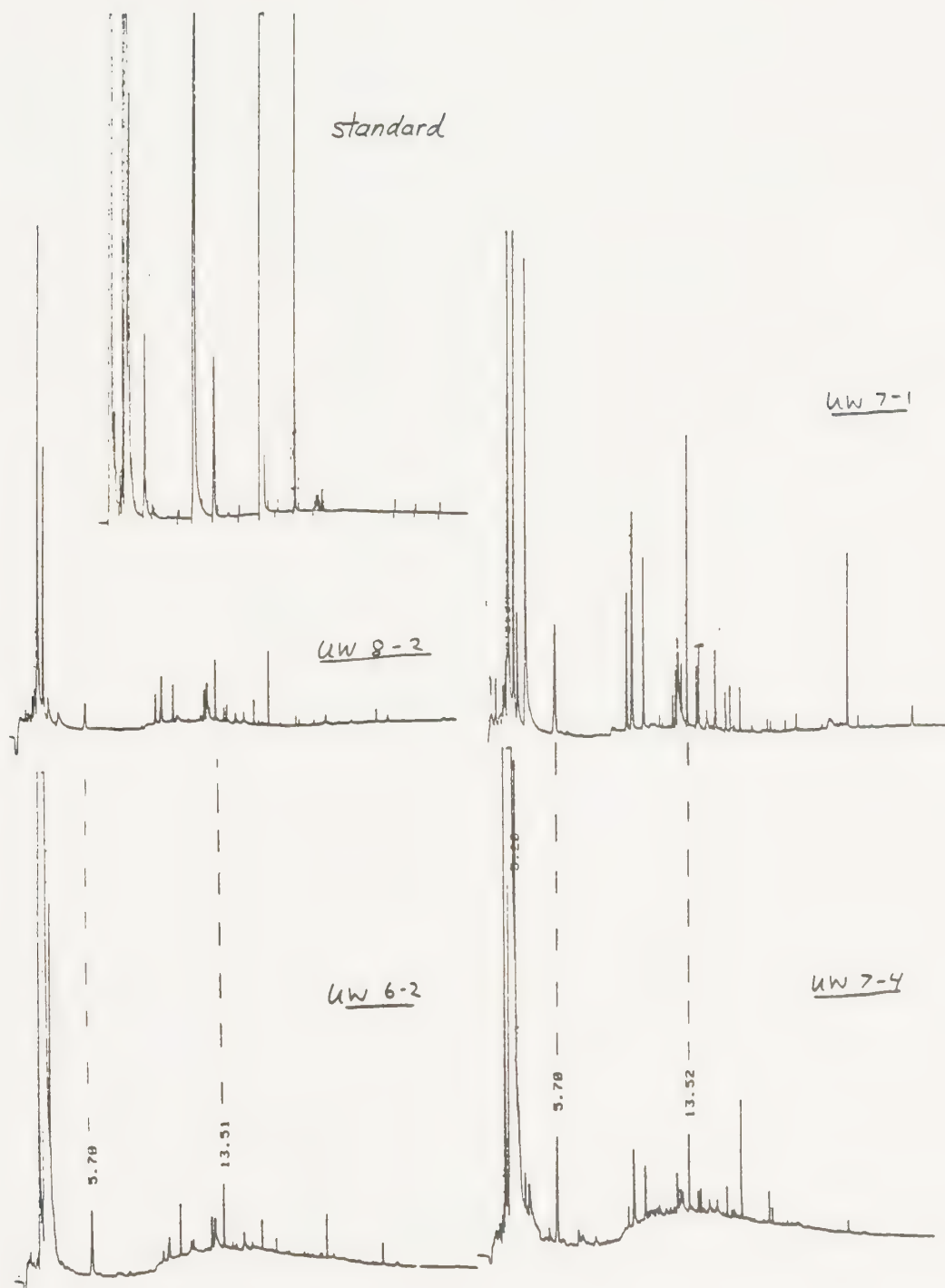


Figure 55. Continued.

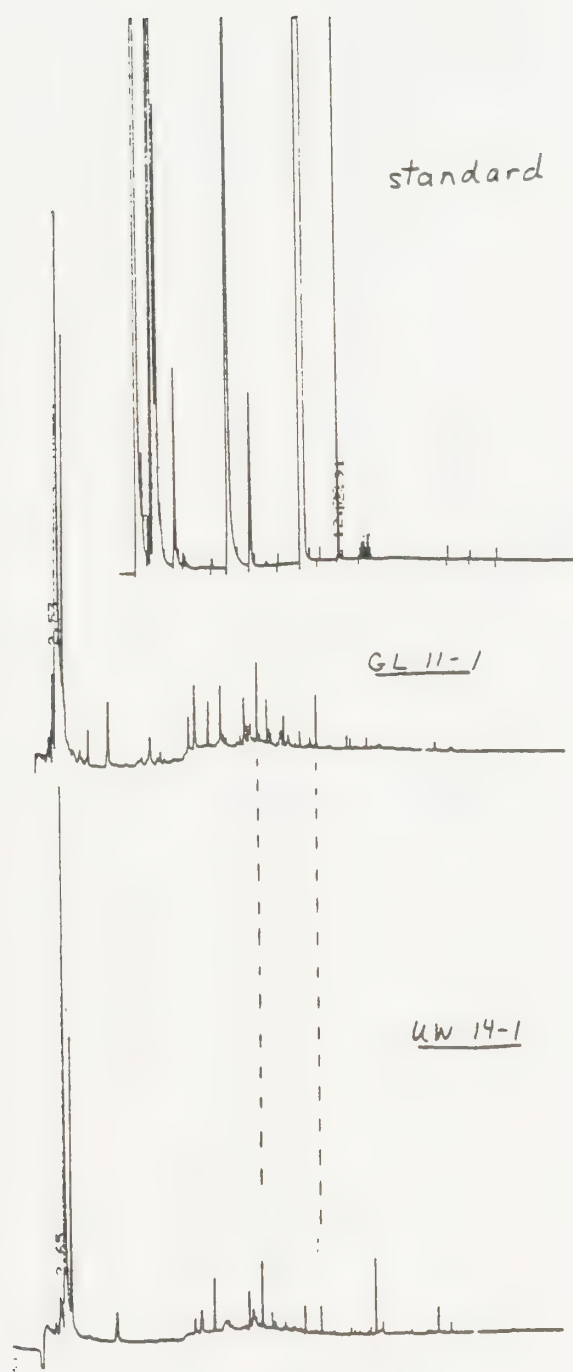


Figure 55. Continued.

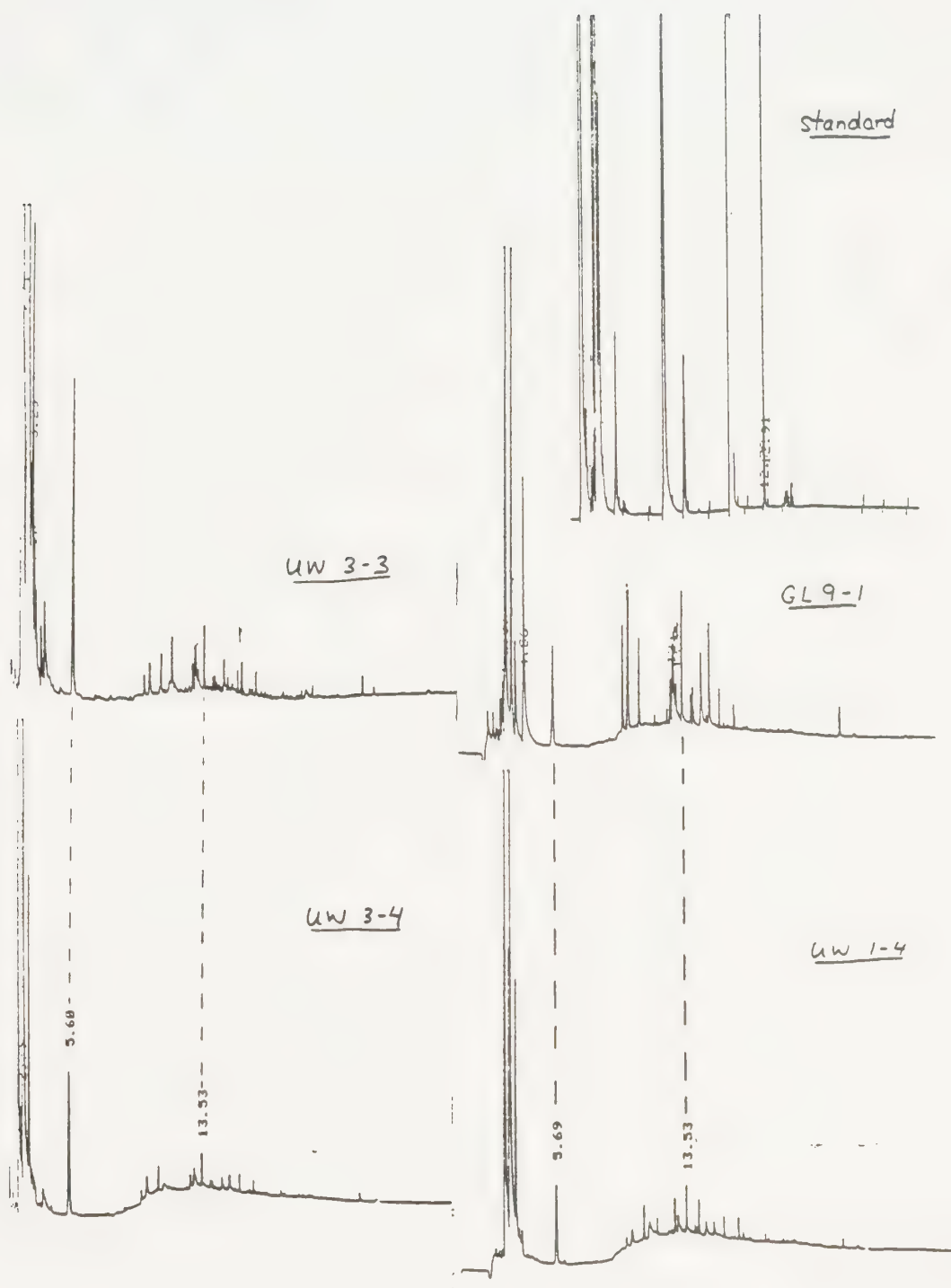


Table 1. Inorganic chemistry of groundwaters near the Upper Ottawa Street Landfill.

Note: Field data was measured on site. Laboratory analyses were performed by the MOE (Rexdale) Laboratory except for TOC which was determined at Waterloo. All concentrations are in mg/l (ppm) except for pH, Eh, (mV), and electrical conductivity (micro Siemens). Uncertain analyses are indicated by ?. Approximate analyses are indicated by ~. Analyses below the detection limit are indicated by <.

Sample	UW 1-2		UW 1-4		UW 2-1	
Date	30/9/82	17/11/82	30/9/82	02/11/82	05/10/82	17/11/82
Field Data						
T, °C	~ 22				18	
Eh, mv	- 79		-145		+33	
pH	6.62		6.64		6.73	
Lab Data						
pH	7.37	7.30	7.16	7.71	7.32	7.19
cond., μS	8980	9320	21300	21900	4330	4170
hardness, as CaCO ₃	1390	2700	6600	5140	1330	1190
Ca	412	810	1850	1390	363	330
Mg	87.0	164	485	406	104	88.0
Na	1000	990	2700	2660	340	255
K	152	162	56.0	60.0	24.0	17.0
Cl	1660	1530	7910	6200	472	576
Alk, as HCO ₃	822	838	183	70.0	331	319
SO ₄		1780		1500		1030
F	0.63	0.70	0.73	0.55	0.44	0.45
NO ₃	0.89	<0.4	0.89		<0.4	<0.4
NO ₂ , as N	0.02	<0.01	0.02	0.03	<0.01	0.03
Kjeld. N, as N	142	89.5?	4.5?		19	12.4?
NH ₃ , as N	133	146	6.8	7.0	18	18.1
TOC	92	100	22	2.7	38	9.2
B	0.60	0.93	3.5	3.6	1.8	2.80
Fe	0.26	0.07	0.068	0.12	0.91	0.05
Mn	0.049	0.014	0.15	0.17	1.2	1.83
As	< 0.030	<0.030	0.050	0.040	< 0.030	< 0.030
Cd	0.0008	<0.002	0.0063	< 0.020	0.011	< 0.002
Cr	0.019	0.020	0.022	0.020	0.017	0.016
Cu	0.045	0.070	0.076	0.110	0.042	0.049
Ni	0.023	0.028	0.005	0.006	0.010	0.011
Pb	0.053	0.050	0.110	0.050	0.027	0.038
Se	< 0.030	< 0.030	< 0.030	< 0.030	< 0.030	< 0.030
Zn	0.013	< 0.010	0.034	0.017	0.120	0.008

Sample	UW 3-3	UW 3-4		UW 6-2
Date	06/10/82	06/10/82	17/11/82	13/10/82
Field Data				
T, °C	19	19		16
Eh, mv	+ 16	+ 16		-167
pH	6.26	6.48		6.94
Lab Data				
pH	7.07	6.94	7.08	7.20
cond., μ S	41100	51600	42300	22100
hardness, as CaCO_3	11900	12100	12600	6080
Ca	3650	3630	3550	1680
Mg	670	745	920	460
Na	6130	7400	9400	2550
K	136	154	150	66
Cl	17600	19000	16600	6790
Alk, as HCO_3	400	350	324	313
SO_4			1340	1340
F				
NO_3	0.4	< 0.4	< 0.4	< 0.4
NO_2 , as N	0.02	0.02	< 0.01	< 0.01
Kjeld. N, as N	~ 34	~ 33	24.5 ?	8.0
NH_3 , as N	35	34.5	44.0	7.2
TOC	65	34	23	19
B	0.52	1.0	5.20	4.2
Fe	0.28	1.4	0.17	
Mn	0.20	0.27	0.194	
As	< 0.030	< 0.030	< 0.030	< 0.030
Cd	0.0016	0.0011	0.002	< 0.020
Cr	0.099	0.085	0.033	
Cu	0.180	0.100	0.170	
Ni	0.023	0.010	0.010	
Pb	0.057	0.054	0.085	< 0.120
Se	< 0.030	< 0.030	< 0.030	< 0.030
Zn	0.640	0.220	0.010	

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Sample	UW 6-4	UW 7-1	UW 7-2
Date	13/10/82	20/10/82	20/10/82
Field Data			
T, °C	14	16	16
Eh, mv	+ 53	- 37	
pH	6.99	6.65	6.89
Lab Data			
pH	6.74	7.35	7.11
cond., μ S	79500	9640	22100
hardness, as CaCO_3	54500	1390	5530
Ca	14100	260	1460
Mg	4700	180	460
Na	24200	1260	3150
K	370	228	112
Cl	71500	1690	7510
Alk, as HCO_3	41.1	752	329
SO_4	835	1500	1740
F		0.67	0.61
NO_3	0.89	0.4	1.3
NO_2 , as N	0.02	0.02	0.02
Kjeld. N, as N	35	190	36.5
NH_3 , as N	28.6	118	27.6
TOC	5.00	140	
B	4.6	1.1	6.0
Fe		< 0.01	0.07
Mn		0.024	0.04
As	< 0.030	< 0.030	< 0.030
Cd	< 0.020	< 0.002	< 0.002
Cr		0.023	0.027
Cu		0.050	0.056
Ni		0.049	0.018
Pb	< 0.120	< 0.012	0.012
Se	< 0.030	< 0.030	< 0.030
Zn		0.120	0.033

Sample	UW 8-2	UW 8-3	UW 9-1
Date	12/10/82	12/10/82	18/11/82
Field Data			X100-101
T, °C	12	14	
Eh, mv	- 95	- 54	
pH	6.55	5.64	
Lab Data			
pH	7.15	6.55	7.62
cond., μ S	6800	73200	7670
hardness, as CaCO_3	2530	28100	2520
Ca	695	7800	725
Mg	194	2100	173
Na	710	10200	840
K	46.0	149	25.5
Cl	1230	37300	1880
Alk, as HCO_3	456	99.4	271
SO_4	1880	1270	1060
F	0.52		0.29
NO_3	0.89	0.89	< 0.4
NO_2 , as N	< 0.01	0.03	< 0.01
Kjeld. N, as N	30.2	34.5	4.0
NH_3 , as N	29.5	33.8	2.8
TOC	22	44	
B	3.40	3.0	2.80
Fe			< 0.03
Mn			0.023
As	< 0.030	< 0.030	< 0.030
Cd		< 0.002	< 0.002
Cr			0.014
Cu			0.060
Ni			0.008
Pb			0.052
Se	< 0.030	< 0.030	< 0.030
Zn			0.009

Sample	UW 9-1		UW 9-2	UW 9-3	
Date	18/11/82	18/11/82	12/10/82	12/10/82	03/11/82
Field Data	X103-104	X105-106			
T, °C			12		
Eh, mv			+ 55	+ 49	
pH			6.82	6.45	
Lab Data					
pH	7.76	7.29	7.40	7.16	6.19
cond., μ S	7660	7660	8810	12600	10900
hardness, as CaCO_3	2530	2340	2330	2400	813
Ca	740	675	650	625	204
Mg	165	165	172	205	74.0
Na	820	850	1290	1820	1660
K	25.0	25.5	38.0	48.0	30.0
Cl	1840	1850	2380	3140	3250
Alk, as HCO_3	269	269	353	393	157
SO_4	1040	1050	1150	515	194
F	0.28	0.30	0.51	0.53	0.42
NO_3	< 0.4	< 0.4	1.3	43	7.5
NO_2 , as N	< 0.01	< 0.01	< 0.01	0.03	< 0.01
Kjeld. N, as N	3.9	3.3	9.4	14.0	11.4
NH_3 , as N	2.7	2.6	9.4	13.6	5.4
TOC			14	2000	2900
B	2.80	2.90	0.55	1.60	2.00
Fe	< 0.02	< 0.03			3.85
Mn	0.021	0.080			0.37
As	< 0.030	< 0.030	< 0.030	< 0.030	< 0.030
Cd	< 0.002	< 0.002	< 0.002		< 0.020
Cr	0.014	0.014			2.000
Cu	0.060	0.060			0.050
Ni	0.011	0.008			0.020
Pb	0.041	0.044	< 0.120		< 0.012
Se	< 0.030	< 0.030	< 0.030	< 0.030	< 0.030
Zn	0.004	0.004			< 0.180

Sample	UW 10-1	UW 13-1	UW 13-5	UW 14-1
Date	12/10/82	19/10/82	19/10/82	21/10/82
Field Data				
T, °C	14			9
Eh, mv	- 09		+ 75	- 136
pH	6.87		7.02	6.82
Lab Data				
pH	7.59		7.07	7.04
cond., μ S	3230		52900	8950
hardness, as CaCO_3	1650		11900	2730
Ca	565		3000	700
Mg	59		1070	240
Na	199		7850	980
K	13.8		182	24.0
Cl	271	1750?	21500	2550
Alk, as HCO_3	168		157	85.8
SO_4	1340		1080	860
F	0.69			0.70
NO_3	< 0.4		1.3	0.89
NO_2 , as N	< 0.01		0.03	< 0.01
Kjeld. N, as N	10.3		28.5	3.7
NH_3 , as N	0.3		25.1	2.0
TOC	8.7	580	620	11
B	0.55		3.30	2.10
Fe			< 0.04	0.28
Mn			1.05	0.30
As	< 0.030		< 0.030	< 0.030
Cd			< 0.002	< 0.002
Cr			0.039	0.011
Cu			0.089	0.035
Ni			< 0.010	< 0.010
Pb			< 0.012	< 0.012
Se	< 0.030		< 0.030	< 0.030
Zn			0.054	0.017

Sample	UW 15-1	UW 15-2
Date	24/9/82	24/9/82
Field Data		
T, °C		
Eh, mv		
pH	6.55	6.75
Lab Data		
pH	7.44	7.45
cond., μS	1250	24800
hardness, as CaCO_3	680	2790
Ca	183	850
Mg	54.6	162
Na	37.2	5500
K	1.90	20.0
Cl	70.0	8730
Alk, as HCO_3	209	169
SO_4	310	780
F	0.65	0.47
NO_3	< 0.4	0.44
NO_2 , as N	< 0.01	< 0.01
Kjeld. N, as N	< 0.2	~ 1.1
NH_3 , as N	0.1	1.5
TOC	15	310
B	0.03	0.33
Fe	0.19	0.002
Mn	0.03	0.001
As	< 0.030	< 0.030
Cd	0.0009	< 0.0004
Cr	0.007	0.002
Cu	0.028	< 0.001
Ni	0.007	0.005
Pb	0.013	< 0.010
Se	< 0.030	< 0.030
Zn	0.027	0.001

Table 2. Values of bulk hydraulic conductivity from water-level recovery data.

Piezometer	Depth (m)	Elevation (m.a.s.l.)	Rock Type	K cm/s	Comments
GLA1- I	11.4	188.6	Shaly Dolomite	4.3×10^{-5}	GLA2 just below landfill recovery too fast to measure GLA4 just below landfill recovery too fast to measure
GLA2- I	22.2	183.0	Cherty Dolomite	1.4×10^{-3}	
- II	22.8	182.5	Cherty Dolomite	high	
GLA4- I	30.4	175.3	Cherty Dolomite	1.4×10^{-3}	
- II	26.1	179.7	Cherty Dolomite	high	recovery too fast to measure
GLA5- I	24.0	176.2	Cherty Dolomite	1.2×10^{-8}	
GLA6- II	17.8	182.4	Cherty Dolomite	8.8×10^{-5}	
GLA6- I	23.5	174.9	Cherty Dolomite	2.9×10^{-6}	
GLA7- I	14.0	183.2	Shaly Dolomite	5.1×10^{-7}	recovery too fast to measure
- II	7.8	189.5	Cherty Dolomite	8.7×10^{-5}	
-III	5.9	191.3	Cherty Dolomite	high	
GLA9- I	10.3	171.9	Cherty Dolomite	1.0×10^{-3}	
GLA11- I	6.7	175.6	Cherty Dolomite	$>1.8 \times 10^{-4}$	recovery too fast to measure
	16.1	167.5	Till	high	
GLA12- I	6.7	177.0	Cherty Dolomite	4.2×10^{-6}	recovery too fast to measure
	3.0	180.6	Shaly Dolomite	4.9×10^{-6}	
GLA12- I	5.8		Cherty Dolomite	high	recovery too fast to measure
				8.6×10^{-4}	
UW 1- 2	4.3	178.1	Cherty Dolomite	$>1.9 \times 10^{-4}$	recovery too fast to measure
- 4	8.1	174.3	Cherty Dolomite	$>1.5 \times 10^{-4}$	
UW 2- 1	4.1	177.3	Cherty Dolomite	$>1.2 \times 10^{-5}$	recovery too fast to measure recovered in < 1 hr.
- 2	8.0	173.5	Cherty Dolomite	$>1.2 \times 10^{-5}$	
- 3	10.1	171.3	Cherty Dolomite	$>1.9 \times 10^{-4}$	recovery too fast to measure
UW 3- 1	4.4	177.1	Cherty Dolomite	$>2.3 \times 10^{-4}$	
- 5	16.3	165.3	Shaly Dolomite	5.0×10^{-9}	recovery too fast to measure
UW 5- 1	7.2	195.5	Shaly Dolomite	$>1.5 \times 10^{-4}$	
- 2	10.5	192.1	Shaly Dolomite	$>1.5 \times 10^{-4}$	recovery too fast to measure
UW 6- 1	3.7	179.8	Cherty Dolomite	$>1.2 \times 10^{-4}$	
- 2	9.0	174.5	Cherty Dolomite	$>1.2 \times 10^{-4}$	recovery too fast to measure
- 5	27.9	155.5	Shaly Dolomite	4.7×10^{-10}	
UW 7- 1	6.4	179.1	Shaly Dolomite	$>1.2 \times 10^{-4}$	recovery too fast to measure
- 2	11.6	174.0	Cherty Dolomite	$>1.2 \times 10^{-4}$	
- 3	17.4	168.2	Cherty Dolomite	$>1.2 \times 10^{-8}$	recovery too fast to measure
- 4	22.3	163.3	Shale	1.9×10^{-9}	
			Shale	1.2×10^{-9}	

Piezometer	Depth (m)	Elevation (m.a.s.l.)	Rock Type	K cm/s	Comments
UW 7-	5	158.1	Limestone & Shale	5.0 x 10 ⁻⁷	
UW 8-	2	179.3	Cherty Dolomite	>1.2 x 10 ⁻⁴	recovery too fast to measure
-	5	160.7	Limestone	6.1 x 10 ⁻⁶	
UW 9-	1	184.0	Shale	>1.2 x 10 ⁻⁴	recovery too fast to measure
-	3	171.2	Shaly Dolomite	1.4 x 10 ⁻⁶	
-	4	163.6	Shale	5.0 x 10 ⁻⁹	
-	5	157.8	Shale	4.2 x 10 ⁻⁸	
UW10-	2	180.9	Shale	1.0 x 10 ⁻⁶	
-	3	176.6	Cherty Dolomite	2.0 x 10 ⁻⁹	
-	5	166.4	Shale	5.8 x 10 ⁻⁶	
UW12-	3	167.8	Shaly Dolomite	5.6 x 10 ⁻⁸	
-	5	157.3	Shale	2.1 x 10 ⁻⁹	
UW13-	2	163.9	Shale	3.5 x 10 ⁻⁵	
-	6	143.4	Shale	2.7 x 10 ⁻¹⁰	
UW14-	1	177.4	Shaly/Cherty Dolomite	>1.2 x 10 ⁻⁴	recovery too fast to measure
UW15-	1	193.4	Shaly Dolomite	>1.2 x 10 ⁻⁴	recovery too fast to measure

Table 3. Concentrations of dissolved methane (CH_4) and total inorganic carbon (TIC) in groundwaters at the Upper Ottawa Street landfill site. 1×10^{-3} mole/l is equivalent to about 22 cm^3 of CH_4 or CO_2 per litre.

Piezometer	Date	Methane, CH_4 , moles/l $\times 10^3$	Total Inorganic Carbon TIC, moles/l $\times 10^3$
UW 1 - 2	30/09/82	0.39	53
	17/11/82	1.0	39
UW 1 - 4	30/09/82	0.36	4.3
UW 2 - 1	05/10/82	0.23	15
	17/11/82	0.37	16
UW 3 - 3	06/10/82	0.08	19
UW 3 - 4	06/10/82	0.66	7.8
	17/11/82	0.97	14
	30/11/82	0.32	16
UW 6 - 2	13/10/82	0.12	10
UW 6 - 4	13/10/82	<0.05	18
UW 7 - 1	20/10/82	1.2	52
UW 7 - 2	20/10/82	0.12	19
UW 8 - 2	12/10/82	1.0	25
UW 8 - 3	12/10/82	0.22	4.0
UW 9 - 2	14/12/82	0.30	18
UW 9 - 3	12/10/82	0.80	14
	05/11/82	0.23	13
	02/12/82	0.14	11
UW 10- 1	12/10/82	<0.05	8.0
	05/11/82	<0.05	7.1
UW 13- 5	19/10/82	0.70	5.5
UW 14- 1	21/10/82	0.26	5.3
	03/12/82	0.19	4.6

Table 4. Concentration of selected, volatile organic compounds in waters near the Upper Ottawa Street landfill site. Results are in $\mu\text{g/l}$ (ppb) and duplicate samples were usually analysed.

Sample	CCl_3	111TCE	CCl_4	BrCl_2M	112TCE	TCEY	CBr_3
UW 3-4	0.02	0.01	-	-	0.01	0.01	0.05
UW 7-1	0.33	0.07	0.02	-	0.06	0.02	-
	0.02	0.01	-	-	0.16	0.01	-
UW10-1	0.02	0.01	-	-	0.004	0.01	0.09
	0.02	0.01	-	-	0.002	0.01	0.01
UW14-1	0.40	0.16	0.04	-	0.004	0.01	0.04
	0.37	0.10	0.02	-	0.004	0.01	-
GL 9-1	0.16	0.23	0.01	-	0.08	0.04	-
	0.18	0.12	0.005	-	0.03	0.01	0.03
Seep near UW 7	0.21	0.08	0.01	-	0.01	0.02	0.29
Leaching Test	0.42	14	-	-	0.03	0.06	0.12
	0.38	12	-	-	0.03	0.04	-

CCl_3 - chloroform, 111TCE - 1,1,1-Trichloroethane

CCl_4 - carbon tetrachloride, BrCl_2M - bromodichloromethane

112TCE - 1,1,2-Trichloroethane, TCEY - Trichloroethylene

CBr_3 - bromoform

Table 5. Environmental isotopes, tritium and $^{18}\text{O}/^{16}\text{O}$, in selected groundwaters. Tritium is reported in Tritium Units and has an uncertainty of ± 10 T.U.. Oxygen isotope ratios are reported as δ values in units of per mil ($^{\circ}/\text{oo}$).

Sample and Date	3T (T.U.)	$\delta^{18}\text{O}$ ($^{\circ}/\text{oo}$)
UW 1-2 30/09/82	120	-10.0
17/11/82		-10.1
UW 1-4 30/09/82	0	- 9.9
02/11/82		- 9.8
UW 2-1 05/10/82	90	- 9.7, -9.8
17/11/82		- 9.6
UW 3-3 06/10/82	30	- 9.8
UW 3-4 06/10/82		- 9.6
17/11/82		- 9.6
30/11/82		-10.2
UW 6-2 13/10/82	20	- 9.8
UW 6-4 13/10/82	20	- 7.6
UW 7-1 20/10/82	110	-10.2
UW 7-2 20/10/82	50	-10.1
UW 8-2 12/10/82	50	- 9.8
UW 8-3 12/10/82	20	-10.5
UW 9-2 12/10/82	50	-10.0
UW 9-3 12/10/82	70	-10.0
03/11/82		- 9.8
UW10-1 12/10/82	60	- 9.5
03/11/82		- 8.8
UW13-1 19/10/82	70	-10.3
UW13-5 19/10/82	20	- 8.1
UW13-6 19/10/82	90	- 8.1
UW14-1 21/10/82	10	- 9.8
30/11/82		- 9.7
UW15-1 24/09/82	70	- 8.5
UW15-2 24/09/82	80	- 9.7
GL 9-1 30/11/82		- 9.8
GL11-1 30/11/82		-10.2
Seepage 01/12/82		-10.1
face near UW7/UW8		

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